

To whom it may concern,

On the 7<sup>th</sup> of March 2021 NRCAN's *Radwaste Review Team* issued a series of four discussion papers on radioactive waste management and requested feedback from interested parties as part of its process of public engagement in the development of a so-called Modernized Radioactive Waste Policy for Canada.

The first of these discussion papers is entitled: *Waste Minimization* and is the topic I wish to address in this email.

First, I consider the *production* of radioactive waste in an operating CANDU reactor. This occurs at a predictable rate that is governed by the neutron flux in the heat transport and moderator systems of each CANDU Unit. These production rates are essentially fixed by the design and materials of construction of the reactor and therefore cannot be modified. In addition, chemical purification of these systems is handled by ion exchange resin columns which also accumulate radioactivity at predictable rates offering no real opportunities for radioactive waste minimization.

Although the concept of waste minimization sounds like a good idea on paper, it is very difficult to implement in practice. The fundamental reason for this is that radioactive waste, once created, cannot be destroyed by simple chemical or physical means although an individual radioactive species will, by its very nature, decay at a predictable rate depending on its half-life.

Thus, it is important to recognize that so-called radioactive waste minimization schemes, such as high temperature incineration of combustible wastes, *do not* reduce the total amount of radioactivity in the waste stream. On the contrary, incineration serves only to *disperse* the radioactivity in a waste by entrainment in the off-gases generated by the combustion process.

This is especially true of volatile radionuclides such as tritium (H-3), carbon-14, Cl-36, I-131 and I-129. Thus, for example, if we combust a bale of radioactive waste containing say 1 Curie of H-3, the remaining ash may well contain only 1 milli-Curie of H-3, but the 0.999 Curies of tritium in the off gases will be released to the environment to the detriment of local residents.

In reviewing the literature on radioactive waste management practices in Canada it is often implied that radioactive waste incineration is a useful approach to radioactive waste minimization. However, as shown below, this is simply not the case.

# CANDU Radioactive Waste and OPG's WWMF

By [REDACTED]

## 1.0 Introduction

We live in a world where many of the things we require to sustain our “modern” life style generate unwanted wastes that have to be “disposed of”. For example, we rely heavily on transportation which requires vast numbers of automobiles, trucks, buses, ships and aircraft which, in turn, rely on mining and extractive metallurgy for their construction. These activities produce copious quantities of wastes such as mine tailings, slag and scrap metal. Furthermore, in the case of modern means of transportation, the manufactured product itself becomes an intractable waste after only a few decades of use.

Another mainstay of modern life is energy production - and especially *electrical* energy – which we happily consume while tolerating the generation of varying amounts and types of waste. However, when it comes to electricity produced from nuclear fuel, the waste takes on a special significance because we are dealing with radioactive waste, something that was not initially present in the fuel but is created by the energy generation process itself. And for all the claimed benefits of nuclear energy, there is no way around the fact that nuclear fission generates vast quantities of highly toxic radioactive waste.

## 2.0 Tritium Production in, and Release from a CANDU Reactor

Canada's unique method of utilizing nuclear fission for electrical energy production – namely the CANDU reactor – not only produces plenty of fission products, (about 1500 Ci/kg initial uranium), but also generates additional radioactive waste through neutron activation of the moderator and primary heat transport D<sub>2</sub>O. The most problematic of these activation products are tritium and carbon-14. Tritium, by virtue of its relatively short half life of 12.3 years, attains an equilibrium activity of about 3,000,000 Ci per Unit after about 10 years of reactor operation. Carbon-14, on the other hand, slowly builds up in the reactor moderator and heat transport system at a rate of about 500 Ci per Unit per year.

A little-known fact about even the best maintained CANDU reactors is how leaky they are. The high pressure (9 MPa), high temperature (290 °C) heat transport system is the principal contributor to D<sub>2</sub>O leakage, most of which occurs through the pressure tube end fittings. Leak rates ~ 10 kg/hour are typical of most of OPG's Units, although about 95 % of the escaping D<sub>2</sub>O is recovered in condensate from vault vapor recovery driers. Thus, PHTS D<sub>2</sub>O, with its

complement of tritium, leaks from an end fitting, flashes to steam, and enters the reactor containment building as water vapour. This tritiated D<sub>2</sub>O vapour quickly achieves a quasi-equilibrium concentration because the activity buildup in containment is countered by two mechanisms: (i) The vapour recovery driers and, (ii) Leakage from containment to ancillary buildings via pathways such as the fuelling machine duct.

The main point of note here is that tritium is produced in vast quantities in a CANDU reactor and, in spite of the reactor operator's best efforts, a significant portion finds its way into accessible areas in the station through chronic leakage pathways. The concentration of tritium in air within a CANDU station is measured in units of  $\mu\text{Ci}/\text{m}^3$  but is usually expressed in terms of the associated derived air concentration limit (DAC) formally known as the maximum permissible concentration in air (MPC<sub>a</sub>), where  $1 \text{ MPC}_a = 10 \mu\text{Ci}/\text{m}^3$ . Airborne tritium concentrations in containment may exceed 100 MPC<sub>a</sub> and air-supplied plastic suits must be worn to enter such high hazard areas. However, routinely accessible areas *outside of containment* such as the Central Services Building also exhibit airborne tritium concentrations up to 0.2 MPC<sub>a</sub>.

Tritium can also escape from a reactor system and become dispersed throughout a station as the result of an accidental spill. For example, in May 2010 a station emergency occurred at Bruce B which was accompanied by the release of 729 Ci of tritium through the Unit 6 Contaminated Stack. The event responsible for this release is documented in SCR 28203235 and its associated Root Cause Investigation Report. At the time of the event, which occurred on May 31<sup>st</sup>, 2010, a Unit 6 planned outage was in progress. During outage work on SDS2 it was observed that moderator D<sub>2</sub>O was being forced out of storage tank 31760-TK1 into the East Delayed Neutron Monitoring Room R-305 where it accumulated on the floor. The active ventilation path from room R-305 eventually lead to airborne tritium levels up to 2000 MPC<sub>a</sub> near air lock No. 2.

It was subsequently estimated that the May 31<sup>st</sup>, 2010, event involved a spill of 240 kg of moderator D<sub>2</sub>O into room R-305. At the time of the spill the tritium content of Unit 6 moderator D<sub>2</sub>O was 20 Ci/kg so that a total of 4800 Ci of tritium was potentially available for release during the event. However, the actual tritium released from the Unit 6 contaminated stack was 729 Ci or 27 TBq equal to about 15 % of the tritium available in the spilt liquid, the remainder being recovered by physical cleanup and/or the vault air driers. Most of the materials used in this type of cleanup operation, such as mop-heads, paper towels, rags, etc. are bagged and shipped to the Bruce Western Waste Management Facility or WWMF.

### 3.0 Tritium in CANDU Waste

As previously noted, a mature CANDU reactor contains about 3,000,000 Ci of tritium. OPG data show that for each of its operating Units, about 10 Ci/hr of tritium escapes from the PHTS and enters containment. Vault driers at Bruce, Pickering and Darlington are specified to reduce the containment building dew point to  $-20\text{ }^{\circ}\text{C}$  which corresponds to about 1000 ppm of water. If this water is 50% reactor  $\text{D}_2\text{O}$  and 50%  $\text{H}_2\text{O}$  from light water ingress into containment, the vault would contain about 50  $\text{MPC}_a$  of tritium which therefore requires plastic suits for entry.

Vapour recovery dryers are also used outside of containment to remove escaped tritiated water that enters auxiliary buildings connected to containment. Accessible areas outside of containment are the reactor Auxiliary Bay and the adjacent Turbine Hall. Station data show that because of the wide-open nature of the powerhouse building, air tends to flow in random directions with considerable back flow leading to tritium in air concentrations that are frequently well above the design target of  $< 0.1\text{ MPC}_a$  at some locations.

As previously noted, operator actions employed to control leaks and spills in a CANDU station lead to the generation of a large volume of cleanup wastes such as paper towels, mops, rags, plastic bags and other assorted containers, protective clothing, booties, etc, all of which are contaminated with tritium. These wastes are classified by OPG as Low Level Waste (LLW) if they exhibit dose rates less than 10 mSv/hr. In addition, wastes with dose rates less than 2 mSv/hr are deemed to be “compactible wastes” while wastes with dose rates less than 0.6 mSv/hr are considered acceptable for incineration if they consist mostly of cellulosic or plastic materials. All other low level wastes are classified as non-processible.

Waste segregation into compactible, incinerable and non-processible categories is carried out at the stations, and the waste is then packaged into clear polyethylene bags in preparation for transportation to the Western Waste Management facility (WWMF) in 1  $\text{m}^3$  Type A metal containers. As might be expected, the tritium content of such waste is highly variable. Thus, tritium activities in the range 0.1 to 1.9  $\text{Ci}/\text{m}^3$  were measured in ~ 1000 waste containers shipped to the WWMF from Pickering and Bruce in the early 1990s. More recent data on the tritium content of incinerable waste processed at OPG’s WWMF are given in Table 1, below, and are consistent with the previously noted measurements. Thus, we see that the tritium activity in incinerable waste received at the WWMF has been in the range 0.1 to 1.9  $\text{Ci}/\text{m}^3$  for over 25 years.

**Table 1: Tritium Activity in WWMF Incinerable Waste in the Period 2007 to 2015**

Year	Incinerated Volume (m <sup>3</sup> )	As-Received H-3 Activity in Waste (Ci/m <sup>3</sup> )
2007	966	0.37
2008	1196	0.61
2009	874	1.53
2010	1332	0.59
2011	1437	0.37
2012	530	0.53
2013	600	0.64
2014	397	0.49
2015	499	0.22

The volume of LLW shipped to the WWMF since its inception in the mid-1970s has varied between 2000 and 6000 m<sup>3</sup> per year, and the current site inventory (2016) now stands at 93,300 m<sup>3</sup>. Similarly, the volume of LLW that has been incinerated at the WWMF has varied between 400 and 2500 m<sup>3</sup>/year, with the highest rates being in the 1970s and early 1980s. This is consistent with the fact that in the period 1975 – 1985 up to 68 % of the waste received at the WWMF was incinerated. However, the LLW incineration rate over the past decade has been much lower – typically only about 20 %.

In a recent report from the DGR Joint Review Panel: *Environmental Assessment Report*, CEAA No. 17520, issued May 2015, we read:

*OPG stated that the primary source of tritium from the DGR Project would be off-gassing from waste packages, particularly non-processible low-level waste in drums and bins, which are not generally sealed air-tight. About 0.4% of the total tritium in the waste would be released per year.*

The issue of tritium escape from stored LLW was also raised in March 2012, well before the CEAA report noted above was issued. This was in an OPG letter CD # 00216-CORR-00531-00108 to Dr. Stella Swanson, Chairperson of the DGR JRP:

*Measurements of fugitive emissions of H-3 from Low Level Storage Buildings (LLSBs) are summarized in the DGR Preliminary Safety Report and have been compared with the tritium inventories in the LLSBs. The average tritium fractional release rate is 0.4%/yr.*

However, the use of the term “fugitive emissions” in this 2012 document may be traced back to as early as 2002, where we find the CNSC stating at a WWMF Licencing Hearing:

*Although the WWMF is designed to prevent and contain uncontrolled dispersion of radioactive or other hazardous substances, there are some fugitive emissions of volatile tritium and carbon-14 escape, mostly from waste storage.*

Thus we see that the CNSC has known, and accepted that the WWMF has been operating with “fugitive” tritium emissions for more than ten years. But what does the word “fugitive” really mean? In the present context, the definition of “fugitive” comes from the 1999 US EPA Code of Federal Regulations in reference to emissions that **do not pass through a stack, chimney or vent; in other words, unmonitored emissions.**

The currently accepted procedure to estimate fugitive tritium emissions from the WWMF is described in a report by N. C. Garisto: *Deep Geologic Repository Pre-Closure Safety Assessment (VI)*, NWMO Report DGR-TR-2009-09, issued August 2009. In Section 3.4.1 of this report we read:

*Airborne tritium emissions from LLW in the DGR during normal operations were estimated based on data on air releases from the same waste packages as stored in the Low Level (waste) Storage Buildings (LLSBs) at the WWMF. These emissions are not usually directly measured and were obtained as described below.*

*OPG (has) provided information on the tritium emissions from LLSBs based on monitoring conducted in 1999 and 2000. The concentration of tritium inside each LLSB was measured at various sampling locations. At the same time, the air flow rates into and out of the LLSBs were measured. The release rate of tritium was calculated by multiplying the actual exhaust airflow rate by the concentration of H-3 in the building and converting to an annual rate.*

*These measurements show that in the period 1999 to 2000 the average total annual tritium emission from the LLSBs was  $8.83 \times 10^{12}$  Bq (239 Curies). Considering that the year 2000 tritium inventory in the LLSBs was  $2.1 \times 10^{15}$  Bq (56,760 Curies), the fractional release rate of H-3 from the LLSBs is 0.004/a.*

The report by Garisto also includes a Table 3.5 which presents estimated tritium inventories in LLW and ILW at the WWMF for the period 1999 to 2007. From this data one may determine the annual fugitive, (i.e. *unreported*), tritium emissions from the WWMF (due to de-gassing of waste) using the equation:

$$\text{Tritium Emissions (Bq or Ci)} = \text{Inventory (Bq or Ci)} \times \text{Release Fraction (RF)}$$

Thus, for the year 2000, we have:

$$\text{Tritium Emissions (Bq or Ci)} = 2.1 \times 10^{15} \text{ Bq (56,760 Curies)} \times 0.004$$

$$\text{Unreported Tritium Emissions} = 8.4 \times 10^{12} \text{ Bq (227 Curies)}$$

Additional, and more recent data for fugitive tritium emissions from the WWMF, (calculated in the same way), and compared to the yearly values for OPG's *reported* tritium emissions from the WWMF over the same time period, are shown in Table 2 below.

**Table 2: WWMF Annual Tritium Emissions: 2007 - 2015**

Year of Operation	WWMF Annual Tritium Emissions (Curies)	
	Reported	Unreported Off-Gassing
2007	362	314
2008	735	324
2009	1338	335
2010	784	346
2011	538	357
2012	281	368
2013	386	378
2014	194	389
2015	112	400

The data in Table 2 clearly show that the *unreported* tritium emissions from the WWMF have been similar to, or have even exceeded the *reported* tritium emissions over the past four years.

But tritium de-gassing from stored LLW is not the only source of unreported tritium emissions from the WWMF because of additional unreported tritium releases from the compaction of polyethylene waste storage bags containing Curie-quantities of tritiated water or water vapor.

Measurements of 59 bags of compacted waste from PNGS and BNGS, (See COG-94-305), showed that 18 of these bags contained tritium in the concentration range 8.4 – 11.3 Ci/m<sup>3</sup>, values well above the 1 Ci/m<sup>3</sup> “average” tritium content that is usually quoted for this type of waste. This labile tritium would have been present as tritiated water vapor occluded in the void spaces of loosely packed LLW.

However, *liquid water* has been reported to be visible inside these bags and some of this would have undoubtedly been expelled during their compaction. In the period 2007 to 2015 an average of 1240 m<sup>3</sup> of compacted LLW was generated at the WWMF *each year* with the compacted material having an estimate density of 1,000 kg/m<sup>3</sup>. If we conservatively assume that the feed waste contained ~ 10 g of 1 Ci/kg tritiated D<sub>2</sub>O per kg of waste, and 1 % of this tritium was released during the compaction process, a total of 124 Ci of tritium would have been released from the Waste Volume Reduction Building (WVRB) each year and contributed to the WWMF’s unreported (fugitive) tritium emissions.

#### 4.0 Carbon-14 in CANDU Waste

The main source of production of C-14 in a CANDU reactor is the (n, α) reaction on O-17 in the moderator D<sub>2</sub>O. (See OPG Report N-REP-034000.1-10001 by K.S. Bagli: “*An Estimate of Carbon-14 Inventory at OPG Nuclear Sites: 1971 – 1998*”).

The activity of C-14, formed in 1 kg of moderator D<sub>2</sub>O after irradiation for a time, t, is given by:

$$A_{14} = N_{17} \sigma \phi \lambda t$$

Where,

N<sub>17</sub> is the number of O-17 atoms in 1 kg of D<sub>2</sub>O

σ is the thermal neutron cross section of the O-17(n,α)C-14 reaction

φ is the thermal neutron flux in the moderator

λ is the radioactive decay constant of C-14 equal to 0.693/t<sub>1/2</sub>, (t<sub>1/2</sub> is the half-life of C-14)

For a one-year irradiation, appropriate values for these quantities are:

$$N_{17} = 1.505 \times 10^{22} \text{ atoms}$$

$$\begin{aligned}\sigma &= 2.35 \times 10^{-25} \text{ cm}^2 \\ \phi &= 1.42 \times 10^{14} \text{ neutrons/cm}^2/\text{s} \\ \lambda &= 0.693/5715 \text{ years}^{-1} \\ t &= 1 \text{ year}\end{aligned}$$

Hence,

$$A_{14} = 6.09 \times 10^7 \text{ Bq/kg}$$

The moderator consists of 305,000 kg of D<sub>2</sub>O, hence the rate of production of C-14 in a typical CANDU Unit is:

$$A_{14} = 18.57 \text{ TBq/Unit/year}$$

C-14 is continuously removed from moderator D<sub>2</sub>O, (as C-14 bicarbonate), by ion exchange (IX) on mixed-bed, (strong-anion/strong-cation), resins in a purification loop. Assuming that absorption on an IX column is the dominant C-14 removal process in a CANDU moderator, C-14 production and removal rates are related by the equation:

$$\frac{dN_{14}}{dt} = R_{14} - \frac{N_{14}}{V} \cdot \varepsilon r$$

Where,

$N_{14}$  is the number of atoms of C-14 in the moderator D<sub>2</sub>O at time  $t$

$R_{14}$  is the rate of production of C-14 atoms

$V$  is the volume of moderator D<sub>2</sub>O

$\varepsilon$  is the efficiency of C-14 removal

$r$  is the volumetric flow rate of the IX purification loop

Solving this equation with the boundary condition  $N_{14} = 0$  when  $t = 0$  shows that the concentration of C-14 in a moderator tends to an equilibrium value,  $N_{14}(\text{eq})$  at large  $t$ :

$$N_{14}(\text{eq}) = \frac{R_{14}V}{\varepsilon r}$$

Rearranging this equation and converting  $N_{14}$  and  $R_{14}$  to activity units (Bq) by multiplying both sides of the equation by the C-14 decay constant  $\lambda$ , leads to a simple expression for the equilibrium concentration, (in Bq/litre), of C-14 in a moderator:

$$[\text{C-14}(\text{eq})] (\text{Bq/l}) = \frac{A_{\text{prod}}}{\varepsilon r}$$

Where  $A_{\text{prod}}$  is the amount of C-14 produced in a moderator in a specified time interval. In the present calculation, a time interval of 1 week will be used, (which corresponds to the airborne emissions measurement interval), in which case the purification flow rate,  $r$ , must be expressed in units of litres per week).

For a typical CANDU Unit, a C-14 production rate of 18.57 TBq/Unit/year has been calculated which is equal to 0.357 TBq/Unit/week. The moderator IX purification flow rate is typically  $\sim 7$  litres/s, or  $4.23 \times 10^6$  litres/week, and the IX C-14 removal efficiency is assumed to be 95% based on estimates in COG Technical Note TN-04-3041: “*Carbon-14 Sources and Release Pathways in CANDU Plants*”, published in Feb 2005.

Hence,

$$[\text{C-14}(\text{eq})] (\text{Bq/l}) = 0.357 \text{ TBq/week} / (0.95 \times 4.23 \times 10^6) \text{ litres/week}$$

$$[\text{C-14}(\text{eq})] = 0.89 \times 10^5 \text{ Bq/litre}$$

The moderator volume is 305,000 litres, hence the average inventory of C-14 dissolved in the moderator is  $2.71 \times 10^{10}$  Bq or 27.1 GBq. By comparison, a moderator IX resin column typically retains up to 10 TBq of C-14 per  $\text{m}^3$  prior to its replacement and each CANDU Unit operating in Ontario generates at least  $5 \text{ m}^3$  of spent ion exchange resin per year, more than half of which is moderator resin.

It has long been recognized that C-14 captured on moderator IX resin will slowly bleed off or be displaced by other compounds during storage or handling. This problem was first recognized at Bruce NGS A and Pickering NGS A in the late 1990s when significant carbon-14 emissions were observed for these stations *even though they were shut down at that time*. Thus, in the period 2000 – 2002, Pickering A’s C-14 releases averaged 0.18 TBq/year, and Bruce A’s C-14 releases averaged of 0.38 TBq/year from stored ion exchange resin wastes in their respective Service Buildings.

Fugitive carbon-14 emissions from the WWMF are discussed in the Report: *Deep Geologic Repository Pre-Closure Safety Assessment (VI)*, NWMO Report DGR-TR-2009-09, issued August 2009. In Section 3.4.1 of this report we read:

*For C-14, the air release rate from the LLSBs was determined based on information collected in 1999 and 2000. Passive samplers were used to measure airborne C-14 release from the exhaust of the LLSBs, and bubblers were used to measure C-14 levels inside the buildings. From these measurements, an upper bound C-14 release from all LLSBs of  $1.53 \times 10^{11}$  Bq/year was estimated.*

Because spent moderator resin is classified as ILW it is expected that the highest fugitive emissions of C-14 originate from this waste which is stored at the WWMF in 12 m<sup>3</sup> or 18 m<sup>3</sup> in-ground containers referred to as IC-12s and IC-18s, respectively. The NWMO Report noted above estimates the airborne C-14 emissions from ICs are up to  $3.55 \times 10^{11}$  Bq/year. These circa 2000 estimates were updated in 2011, as described in the DGR Project Information Request EIS-06-244 where we read:

*The maximum airborne C-14 release rate from all LLW and ILW packages is  $1.9 \times 10^{12}$  Bq (51 Ci) per year.*

Thus we see that the unmonitored release of C-14 is much larger, (by a factor of about 500) than the monitored release which is typically  $\sim (3.7 \pm 2.3) \times 10^9$  Bq (0.1 Ci) per year.

## 5.0 Radio Iodine in CANDU Waste

Radio iodine, mostly as short-lived, ( $t_{1/2} \sim 8$  days), I-131, is a high-yield fission product that accumulates in CANDU fuel bundles throughout their approximately 1-year residence in a reactor core. In the event of a fuel defect, high-pressure D<sub>2</sub>O penetrates the Zircaloy cladding and directly contacts the irradiated UO<sub>2</sub> resulting in the release of I-131 to the primary heat transport system (PHTS). Gamma-spectrometry of grab samples of PHTS D<sub>2</sub>O readily detects the presence of I-131 and is used on a routine (daily) basis to monitor the concentration of this species in a Unit's heat transport system

As we have seen for the case of tritium, there is always some degree of chronic leakage of heat transport D<sub>2</sub>O into the reactor vault and because of the solubility and volatility of radioiodine, this species is carried over with the leaking D<sub>2</sub>O. As a result, radioiodine is also present in radioactive waste containing materials such as paper, cloth or mop-heads that were used to clean up PHTS D<sub>2</sub>O spills or surfaces that were contaminated with heat transport D<sub>2</sub>O.

A survey of radiochemical data for operating CANDU Units in Ontario shows that I-131 is present in PHTS D<sub>2</sub>O at a relatively constant "background" level. Of course, in the event of a major fuel failure, the I-131 activity may increase to over 100 times the background level; but for a simplified calculation we shall consider that 0.15  $\mu$ Ci of I-131 per kg of D<sub>2</sub>O is the long-term average concentration of radioiodine in a CANDU PHTS. At the same time the concentration of tritium in PHTS D<sub>2</sub>O is practically constant at  $1.0 \pm 0.1$  Ci/kg D<sub>2</sub>O. Thus, the ratio of I-131 to H-3 in heat transport water is:

$$[\text{I-131}]_{\text{PHTS}} / [\text{H-3}]_{\text{PHTS}} = 0.15 \times 10^{-6}$$

In the absence of radioactive decay, this ratio should be maintained in the airborne radioactive emissions from the WWMF incinerator. Conversely, any deviation from this ratio may be attributed to the effects of I-131 decay in the time period between the waste being generated at a nuclear plant and the waste being incinerated at the WWMF. Radioactive emission data reported for the WWMF show that in the time period 2007 - 2015:

$$[I-131]_{\text{WWMF}} / [H-3]_{\text{WWMF}} = 0.38 \times 10^{-8}$$

To reconcile these ratios, we need an I-131 decay factor of 39.5, which corresponds to a delay of about 40 days. This implies that radioiodine contaminated waste takes about 5 weeks to get from its source, (a CANDU reactor PHTS), to its disposal site, the WWMF. This 5 weeks would be the sum of handling time at the station, plus shipping time, plus handling time at the WWMF. Once at the WWMF, about two thirds of the radioiodine is sent for permanent storage, while the remaining third is in “combustible waste” and is released to the environment via the WWMF stack upon incineration of this waste.

However, radioiodine can be released not only by waste incineration, but by escape from the ion exchange resin used to remove it from the coolant. This escape is promoted by heating effects from self-radiation. The volatility of radioiodine in samples of spent resin from Pickering was studied in the 1990s, (See for example AECL-10289) and showed that up to 0.35 % of the I-131 in a sample of spent resin is volatilized within 40 days of removal from a reactor.

If we conservatively assume that 20 m<sup>3</sup> of spent moderator resin is shipped to the WWMF each year and the as-received I-131 activity is 2 Ci/m<sup>3</sup>, this would have decayed to 1.25 Ci after 40 days of storage. Then, if 0.35 % of the available I-131 is released as described above, we have a total annual emission equal to 0.035% × 1.25 Ci or 4.375 mCi, (1.62 × 10<sup>5</sup> Bq), of I-131. This *unmonitored* release is almost twice the annual monitored stack release of radioiodine from the WWMF, which is typically about (9 ± 3) × 10<sup>4</sup> Bq.

Finally, it is worth noting that I-131 activities comparable to, or even greater than those of long-lived Cs-137 have occasionally been measured on WWMF incinerator stack filters (by gamma spectrometry) suggesting that I-131 emissions from waste incineration are highly variable and are therefore quite unpredictable.

## **6.0 Radio Cesium in CANDU Waste**

In the present context, we take the term “radio cesium” to mean the three long-lived fission products: Cs-134, Cs-135 and Cs-137. Of these, Cs-137 is by far the most important and

generally accounts for more than 80 % of the radioactive cesium in CANDU waste. Hence, in this discussion, we will focus our attention on Cs-137. In addition, being a volatile fission product, Cs-137 behaves a lot like I-131 (See Section 5.0), and is present in PHTS D<sub>2</sub>O at a relatively constant “background” level of ~0.2 µCi/kg. At the same time, the concentration of tritium in PHTS D<sub>2</sub>O is practically constant at 1.0 ± 0.1 Ci/kg D<sub>2</sub>O. Thus, the ratio of Cs-137 to H-3 in primary heat transport, (PHTS), water is:

$$[\text{Cs-137}]_{\text{PHTS}} / [\text{H-3}]_{\text{PHTS}} = 0.2 \times 10^{-6}$$

Cs-137’s half-life is about three times longer than tritium’s and tritium is more volatile than Cs-137, therefore Cs-137 tends to become enriched relative to tritium in “old waste”. This is seen in the [Cs-137] / [H-3] ratio in LLW stored at the WWMF which is ~ 3.5 × 10<sup>-3</sup> – a value that is more than 1000 times *larger* than the PHTS ratio noted above.

These relatively high ratios serve to emphasize that Cs-137 is a persistent radionuclide that needs to be reliably tracked in nuclear waste. This is why it is important to quantify the *loss* of Cs-137 from the WWMF during LLW incineration. This may be accomplished using data in OPG’s *DGR Reference Inventory* report. Thus, the current inventory of Cs-137 in LLW waste at the WWMF is estimated to be 1.2 × 10<sup>13</sup> Bq of which only about 1% is found in waste incinerator ash. This suggests that a substantial fraction of the Cs-137 in LLW is volatilized during the combustion process and potentially released to the environment.

It is instructive to compare the behavior of Cs-137 to the other major long-lived beta-emitting fission product in OPG’s LLW: namely, Sr-90. To make this comparison we have carried out activity balance calculations for Sr-90 and Cs-137 as follows:

First, we calculate the total activity in the feed waste and the incinerator ash using specific activity values  $S_x(m)$ , for species  $x$  in medium  $m$ , in the two combustible forms of waste, (bales and boxes), and the two types of ash, (bottom ash and baghouse ash), as listed in OPG’s *DGR Reference Inventory* report and summarized in Table 3 below. Also included in Table 3 are values of a quantity we referred to as an “enrichment factor” which is defined as the ratio of specific activity of the radionuclide in the incinerator ash to its specific activity in the feed.

**Table 3: Specific Activities and Enrichment Factors for LLW Incinerated at the WWMF**

Radionuclide	Specific Activity, $S_x$ , of Radionuclide $x$ (Bq/m <sup>3</sup> )						Enrichment Factor
	Bottom Ash	Baghouse Ash	Ash Average	Bales	Boxes	Feed Average	
<b>Sr-90</b>	$8.2 \times 10^7$	$3.0 \times 10^7$	<b><math>5.6 \times 10^7</math></b>	$3.0 \times 10^6$	$3.6 \times 10^6$	<b><math>3.3 \times 10^6</math></b>	<b>17.0</b>
<b>Cs-137</b>	$5.6 \times 10^7$	$5.0 \times 10^7$	<b><math>5.3 \times 10^7</math></b>	$6.2 \times 10^7$	$7.6 \times 10^7$	<b><math>6.9 \times 10^7</math></b>	<b>0.83</b>

Next, we use the relation:

$$\text{Activity of } x \text{ incinerated per year} = \text{Volume incinerated} \times S_x(\text{feed})$$

Then, if we take 1000 m<sup>3</sup> as the average volume of LLW incinerated each year, we have the following activity balance equation:

$$1000 (\text{m}^3) \times S_x (\text{feed}) (\text{Bq/m}^3) = 1000 (\text{m}^3) \times S_x (\text{ash}) (\text{Bq/m}^3) / V + A_x (\text{Bq})$$

Where,

$A_x$  (Bq) is the activity of species  $x$  released per year

$V$  is the volume reduction factor for the incinerator

From the above equation we obtain by rearrangement the relation:

$$A_x (\text{Bq}) = 1000 (\text{m}^3) \times S_x (\text{feed}) (\text{Bq/m}^3) - 1000 (\text{m}^3) S_x (\text{ash}) (\text{Bq/m}^3) / V$$

Thus, for example, if we assume that the incinerator volume reduction factor,  $V$ , is 20, and substitute the appropriate specific activity values from Table 3, we find for the radionuclides of interest:

$$A_{\text{Cs-137}} = 6.6 \times 10^{10} \text{ Bq}$$

$$A_{\text{Sr-90}} = 5.0 \times 10^8 \text{ Bq}$$

This result is consistent with the enrichment factors given in Table 3 which show that Sr-90 is enriched in LLW incinerator ash while Cs-137 is depleted. This difference is due to the

significant difference in the boiling points of these species: 1382 °C for Sr and 671 °C for Cs. This results in the volatilization of Cs-137 and retention of Sr-90 during LLW incineration.

While these calculations suggest that a considerable amount of Cs-137 is being released by incineration of LLW – namely  $6.6 \times 10^{10}$  Bq/yr – it remains to be determined how much of this, if any, escapes into the environment. Certainly, OPG claim that the WWMF incinerator has particulate filters rated at an efficiency of 99.97% for particle sizes  $\geq 1 \mu\text{m}$ . Nevertheless, this means the incinerator still releases 0.03 % of its waste combustion particulate to the atmosphere, which amounts to  $2 \times 10^7$  Bq of Cs-137 per year or about 100 times the reported annual particulate emissions for the WWMF. This discrepancy is no doubt mainly due to the fact that Cs-137 in waste incinerator flue gas is present as very fine “ nano-particles” which are not captured by the incinerator filters and therefore escape into the environment.

In Bruce Power’s *Annual Summary and Assessment of Environmental Data for 2006*, (See Report: B-REP-03419-00007, issued April 2007), we read:

*Samples of garden soil at location BR2 and BR5 were obtained from the Scott Point area north of the Bruce Power site. Cesium-137 levels were measured at 7.88 Bq/kg and 6.40 Bq/kg which are slightly above the provincial background levels and can be attributed to activities at the Bruce Power site.*

By comparison, OPG have reported levels of Cs-137 up to 29.2 Bq/kg in soil collected about 100 meters to the south of the WWMF confirming that the waste incinerator is responsible for substantial deposition of Cs-137, both on and off the Bruce Nuclear site, (See OPG Report: *Environmental Risk Assessment for the Western Waste Management Facility*, report No. 01098-REP-07701-00011-R000).

## **7.0 Alpha-Emitters in CANDU Waste**

Alpha-emitting radionuclides such as Pu-238, Pu-239, Pu-240, Am-241 and Cm-244 are produced in significant quantities in CANDU fuel and enter an operating Unit’s PHTS in the event of a fuel bundle defect or in-core damage. When a reactor is opened for maintenance operations, or in the event of a spill of PHTS D<sub>2</sub>O, alpha contamination may spread into the surrounding area and be picked-up by mops, paper towels, etc, during clean-up operations. Thus, even so-called Low Level Waste may contain significant amounts of alpha-emitting radionuclides.

The level of alpha contamination that may occur in a CANDU vault may be determined by taking smears of surfaces of interest and measuring the collected activity with a dual-channel scintillation counter such as a Ludlum Model 3030 to evaluate the gross alpha and gross beta activity on the sample using two separate counting periods of a few minutes each. The average ratio of gross beta to gross alpha activity measured on a set of smears is also used for comparative purposes to identify the overall level of alpha contamination in a work area – ratios less than about 50 being considered high alpha-hazard areas. A typical set of such data is presented in Table 4, below, and shows how ubiquitous alpha-contamination is in accessible areas of a CANDU station.

**Table 4: Gross Beta/Alpha Data for Smears Collected at Various Locations in the Bruce Unit 1 Vault April – May 2010**

Sample Description	Gross Beta (Bq)	Gross Alpha (Bq)	Beta/Alpha Ratio
Boiler Alley E	235	17	14
Feeder Cab E FCE-1-6	664	26	26
Preheater Inside HX3-1	2880	100	29
Preheater Inside HX3-2	11900	509	23
Preheater Inside HX3-3	8700	222	39
HX4 Preheater	183	21	9
Bleed Condenser	31	1	31
HX1 Preheater	274	39	7
HX2 Preheater	568	79	7
Feeder Cab W FWCW-1-3	3340	354	9
Feeder Cab WFCW-4-6	3090	320	10
Cable Tray	659	105	6
Boiler Alley W	738	52	14
Pressurizer	91	10	9
Bleed Cooler	105	10	11
Booster Pit	155	16	10
Fan Room	2980	85	35

The reason that alpha-contamination is of such great concern is that the radiation dose per Becquerel from an alpha-emitting radionuclide is much higher than the equivalent dose per Becquerel from a beta-emitter. Thus, the dose conversion factors for beta-emitters such as Co-60 or Cs-137 are  $\sim 3.0 \times 10^{-8}$  Sv/Bq, compared to the dose conversion factors for alpha emitters such as Pu-239 or Am-241 which are  $\sim 1.5 \times 10^{-5}$  Sv/Bq, or about 500 times higher.

OPG measures gross beta particulate in the WWMF's incinerator stack emissions on a regular basis, but does not routinely measure gross alpha-emitting radionuclides in its airborne emissions. Nevertheless, incinerator beta/alpha ratio data has been collected intermittently over the past decade, (See, for example, COG-01-087). These data show a great deal of scatter, but exhibit an average beta/alpha ratio of  $107 \pm 67$ . Thus we see that the alpha emissions from the WWMF, which are not routinely monitored, are 5 times more hazardous than the beta emissions which are routinely monitored.

## **8.0 Radioactive Waterborne Emissions from the WWMF**

So far in our discussion of radioactive releases from the WWMF we have focussed on airborne emissions, but it is important to also consider waterborne emissions from this facility. OPG monitors waterborne emissions from the WWMF via the storm water runoff and subsurface drainage systems. The only radionuclides detected in this water are tritium and gross beta, both of which have been steadily increasing over the past four years and now stand at  $4.28 \times 10^{11}$  Bq (11.6 Ci) per year for tritium and  $1.56 \times 10^8$  Bq (0.0042 Ci) per year for gross beta.

However, as with the WWMF's airborne emissions, there are significant *unmonitored* waterborne radionuclide emissions from this facility. Principal among these emissions is the significant amounts of tritium in the LLSB and IC sump water which is sent to the Bruce water treatment plant and subsequently discharged into the Baie du Doré at the north end of the Bruce Power site; *these releases are not included in the WWMF annual emissions data*.

Nevertheless, the quantification of these fugitive waterborne tritium emissions from the WWMF is discussed in the previously noted report: *Deep Geologic Repository Pre-Closure Safety Assessment (VI)*, NWMO Report DGR-TR-2009-09, issued August 2009. Based on the data presented in Section 3.4.2 of this NWMO report, about 50 m<sup>3</sup> of sump water is collected per year containing an average of  $1.26 \times 10^{10}$  Bq/m<sup>3</sup> of tritium. Hence, we conclude that there are about  $6.3 \times 10^{11}$  Bq (17.0 Ci) of *unmonitored* waterborne tritium releases from the WWMF per year. This is greater by a factor of about 1.5 than the *monitored* waterborne tritium emissions of  $4.28 \times 10^{11}$  Bq (11.6 Ci) per year.

## 9.0 Discussion

Clearly, the most striking feature of the data presented thus far in this report is that the WWMF's airborne and waterborne radioactive emissions are seriously under-reported. The most significant discrepancy in this regard is for airborne carbon-14 where the unmonitored emissions are about 500 times higher than the monitored (reported) emissions.

Of course, OPG and the CNSC's predictable response to this type of anomaly is to claim that the WWMF's radiological emissions are so far below the mandated DRLs, (Derived Release Limits), that even a many-fold increase in their magnitude would still keep them well within allowed limits. However, before accepting this argument we need to look more closely at the DRLs currently in effect for the WWMF and consider how these DRLs are determined.

The official DRLs, (i.e. the values accepted by the CNSC), for airborne and waterborne tritium and carbon-14 emissions from the WWMF are as shown in Table 4 which also includes the WWMF's current inventories of these species:

**Table 4: Tritium, Carbon-14, I-131 and Beta-Particulate DRLs for the WWMF**

Radionuclide	Inventory (Bq)	DRL (Bq/yr)	
		Airborne	Waterborne
Tritium	$3.6 \times 10^{15}$	$2.96 \times 10^{17}$	$7.7 \times 10^{15}$
Carbon-14	$3.3 \times 10^{15}$	$1.09 \times 10^{15}$	-
I-131	-	$1.90 \times 10^{12}$	-
Beta-Particulate <sup>1</sup>	$2.0 \times 10^{13}$	$2.34 \times 10^{12}$	$4.56 \times 10^{11}$

Ref 1: Taken to be Cs-137 in LLW

The first point to note about the data in Table 4 is that the tritium DRL's, both for airborne and waterborne emissions, are *larger* than the WWMF's entire tritium inventory. This leads to the absurd situation that OPG could incinerate and/or flush *all* of its tritiated waste into Lake Huron without exceeding any statutory limit, and therefore (according to the CNSC) do this without causing any harm to members of the public living nearby. This is simply not true because the

methodology used to calculate DRLs grossly *underestimates* radiation doses to exposed individuals. To understand why, we need to first review how DRL's are determined.

By definition, a DRL is a *calculated* radionuclide release rate that, if maintained over a one-year interval, would cause an exposed individual to receive a radiation dose equal to the regulatory annual dose limit, which is currently set at 1 mSv. Furthermore, DRL calculations assume that radioactive species from the facility in question are released at a rate which is more or less constant and occurs via a *monitored* stack or liquid effluent stream. Thus, there are two DRL's that are calculated for a given radionuclide, one for airborne emissions, and another for waterborne emissions.

An important feature of the DRLs for a nuclear facility – whether it's a power station with large radiological emissions or a nuclear waste disposal site with much lower emissions – is that a DRL is essentially *independent* of how a radioactive emission is produced, but *strongly dependent* on the degree of exposure of an individual to the emission. This is because the impact of a radiological exposure varies according to such factors as the proximity of the individual to the source of the release, his or her age, lifestyle, dietary habits, etc. Thus, for example, a farmer who drinks his own well-water and eats his own produce is likely to receive a higher dose from tritium emissions from an adjacent nuclear facility than a non-farming resident who lives near a nuclear facility but drinks bottled water and eats imported food.

In developing a set of DRLs for a nuclear facility it is therefore standard practice to classify exposed individuals as belonging to groups with shared exposure factors. The group that is predicted to receive the *highest* dose from a radionuclide is referred to as the critical group. For the Bruce nuclear site, two critical groups are recognized: agricultural farm residents located near Inverhuron Provincial Park to the south of the site, designated BF14; non-farm residents near Scott Point located on the shore of Lake Huron northeast of the nuclear site, designated BR1.

An interesting feature of the DRLs for the WWMF is that this facility is located near the center of the Bruce Power nuclear site which it shares with Bruce A, Bruce B and the Central Maintenance and Laundry Facility, (CMLF). For three of these facilities – the WWMF, the CMLF and Bruce B – the critical group for the determination of the DRL for airborne tritium emissions is the same: “Bruce Farmer BF14” who resides a few km south of the site. For this reason, we would

expect the airborne tritium DRLs for the WWMF, the CMLF and Bruce B to be essentially the same. This is discussed further below, but first we need to consider how inhalation doses are calculated.

The inhalation dose *per year* from the release of a radioactive species such as tritium is given by:

**Inhalation Dose (Sv/year) =**

$$\text{Release (Bq/year)} \times \text{DCF (Sv/Bq)} \times \text{BR(m}^3\text{/s)} \times \text{(X/Q) (s/m}^3\text{)} \dots \text{Eqn. 1}$$

Here *DCF* is the dose conversion factor (in Sv/Bq), **BR** is the receptor's breathing rate (in m<sup>3</sup>/s), **Q** is the release rate of the radionuclide (in Bq/s), and **X** is the concentration (in Bq/m<sup>3</sup>). The **(X/Q)** term is an atmospheric dispersion factor that drops off rapidly with distance from an emission source. Thus, for example, at a station's "regulatory site boundary", generally taken to be about 1 km from the emission source, **(X/Q)** is in the range 10<sup>-6</sup> to 10<sup>-5</sup> s/m<sup>3</sup>, while at 10 km from the site boundary **(X/Q)** is generally less than 10<sup>-7</sup> s/m<sup>3</sup>. However, the precise value of an atmospheric dispersion factor depends on the time scale of the release.

For short-term releases the maximum downwind concentration of a radionuclide of interest occurs along the plume centerline and the atmospheric dispersion factor is given by:

$$\text{(X/Q)}_{\text{short}} \text{ (s/m}^3\text{)} = \text{Exp}\{-\text{h}^2/2 \sigma_z^2\} / (\pi\sigma_y\sigma_z\text{u}) \dots \text{Eqn. 2}$$

Where,

**σ<sub>y</sub>** is the standard deviation in the concentration in the crosswind direction in meters

**σ<sub>z</sub>** is the standard deviation in the concentration in the vertical direction in meters

**u** is the average wind speed in meters per second

**h** is the stack height in meters

For long-term releases the wind does not always blow with the same speed or in the same direction so that a correction factor must be applied to the short-term release equation to allow for these variabilities. This is accomplished by dividing

all possible wind directions into  $n$  sectors and replacing  $\sigma_y$  in the short-term release equation by the sector width at a distance  $x$  from the emission source, or  $2\pi x$  divided by the number of sectors. This is typically taken to be the sixteen  $22.5^\circ$  compass directions – N, NNE, NE, etc. In addition, the concentration in each sector is weighted by the fraction of time,  $f_i$ , that the wind blows into sector  $i$ . And this is further weighted by a factor  $F_{jk}$  which is the fraction of time during which a Pasquill-Gifford atmospheric stability class,  $j$ , is observed for a wind class  $k$ . The resulting, so-called “triple-frequency”, atmospheric dispersion factor is given by:

$$(X/Q)_{\text{long}} \text{ (s/m}^3\text{)} = 2.032 f_i / x \cdot \sum_{jk} F_{jk} / (u_k \sigma_{zj}) \cdot \text{Exp}\{-h^2/2 \sigma_{zj}^2\} \dots \text{Eqn. 3}$$

Frequencies of occurrence of the atmospheric stability classes, mean wind speeds and wind directions for the Bruce, Pickering and Darlington NPPs have been published in the AECL report: S. L. Chouhan et al. “*Testing the Atmospheric Dispersion Model of CSA N288.1 with Site-Specific Data*”, AECL Report No: AECL -12099, January 2001. These frequencies may be used to calculate typical long-term atmospheric dispersion factors for the hypothetical case of a critical group located  $x$  km from a ground level source.

### 9.1 Tritium Emissions

Returning to the question of airborne tritium DRLs for the WWMF, this quantity may be evaluated by rewriting Eqn. 1 with the dose set to 1 mSv and the time interval to 1 year as required by CSA Standard N288.1:

$$DRL(\text{Bq/yr}) = 10^{-3}(\text{Sv})/[DCF(\text{Sv/Bq}) \times BR(\text{m}^3/\text{s}) \times 3.15 \times 10^7(\text{s/yr}) \times (X/Q)(\text{s/m}^3)]$$

For the case of a long-term exposure, with a relatively constant tritium emission rate, Eqn. 3 has been evaluated for a receptor located 1 km from the Bruce center of site in which case  $(X/Q)$  is  $1.58 \times 10^{-6} \text{ s/m}^3$ , and the tritium DRL is equal to  $1.05 \times 10^{17} \text{ Bq}$ . However, the receptor of interest is actually located at BF14 and the distances from the three major tritium sources on the Bruce site – the WWMF, the CMLF and Bruce B – are as follows:

$$\text{WWMF} = 2.813 \text{ km}$$

$$\text{CMLF} = 2.350 \text{ km}$$

$$\text{Bruce B} = 2.933 \text{ km}$$

Then using the relation:

$$(X/Q) (x \text{ km}) = (X/Q) (1 \text{ km}) / x,$$

the (X/Q) values appropriate to a receptor at BF14 are as follows:

$$(X/Q) \text{ for the WWMF} = 5.617 \times 10^{-7} \text{ s/m}^3$$

$$(X/Q) \text{ for the CMLF} = 6.723 \times 10^{-7} \text{ s/m}^3$$

$$(X/Q) \text{ for Bruce B} = 5.387 \times 10^{-7} \text{ s/m}^3$$

Table 5, below, presents the airborne tritium DRLs calculated from these (X/Q)'s and compares the resulting values to the DRLs currently accepted by the CNSC for these facilities. Clearly, the agreement between the two sets of values is excellent.

**Table 5: Comparison of Calculated and CNSC-Approved DRLs for Airborne Tritium**

Facility	Calculated DRL (Bq)	CNSC Approved DRL (Bq)
WWMF	$2.95 \times 10^{17}$	$2.96 \times 10^{17}$
CMLF	$2.47 \times 10^{17}$	$2.45 \times 10^{17}$
Bruce B	$3.08 \times 10^{17}$	$3.16 \times 10^{17}$

It is also instructive to calculate the tritium in air concentrations at BF14 predicted by the (X/Q) values noted above for the three nuclear facilities of interest. To this end, we have calculated the tritium in air contribution from each facility and compare the *derived* values to the *measured* values at BF14. This calculation was accomplished using the formula:

$$\text{Tritium in Air Concentration at BF14 (Bq/m}^3\text{)} =$$

$$Q_{\text{WWMF}} \times (X/Q)_{\text{WWMF}} + Q_{\text{CMLF}} \times (X/Q)_{\text{CMLF}} + Q_{\text{BB}} \times (X/Q)_{\text{BB}}$$

Where,

$Q_F$  is the tritium release rate, (in Bq/s), from facility F

$Q_F$  values for each facility vary from year to year and are published in Bruce Power's *Annual Summary and Assessment of Environmental Radiological Data* reports. We have taken data from 2006 to 2011 to determine the tritium concentrations at BF14 and compared the resulting values to measured concentrations at this site, as summarized in Table 6 below.

**Table 6: Calculated and Measured Tritium in Air Concentrations at BF14:  
Values for the period 2006 to 2011**

Date	BB Release Rate (kBq/s)	CLMF Release Rate (kBq/s)	WWMF Release Rate (kBq/s)	Calculated Concentration (Bq/m <sup>3</sup> )	Measured Concentration (Bq/m <sup>3</sup> )
2006	14,100	0.413	1,740	8.56	1.31
2007	19,900	0.381	425	10.95	2.20
2008	14,300	0.429	863	8.22	2.19
2009	14,700	0.281	1,570	8.81	1.84
2010	29,000	0.381	921	16.16	2.59
2011	22,800	0.310	632	12.62	1.44

As previously noted, the measured tritium concentration data in Table 6 are taken from Bruce Power's *Annual Summary and Assessment of Environmental Radiological Data* reports for the years in question. These data are derived from passive tritium monitors that were employed at BF14 up to 2011. However, in 2010 there were a number of investigations into the efficacy of passive tritium monitors and it is now recognized that such monitors provide unreliable data. Passive tritium monitors are therefore no longer used at the BF14 site and most regrettably, it appears that no alternative tritium monitors have been installed to replace them.

It is also worth noting with regard to the tritium in air data shown in Table 4 that questions about the accuracy of such data were raised as early as 2001 in a study by S.L. Chouhan et al. entitled: *Testing the Atmospheric Dispersion Model of CSA N288.1 with Site-Specific Data*, AECL Report No. AECL-12099, issued January 2001. This AECL study used tritium-in-air monitoring data recorded at 10 locations in the vicinity of the Bruce site and compared the

reported values to calculated values using the CSA N288.1 recommended methodology. The magnitude of the differences between the predicted and measured tritium concentrations led the authors of the AECL study report to conclude:

*“Large discrepancies between predictions and observations at specific monitoring sites suggests that it is the measurements rather than the model that are at fault.”*

What is most interesting about the suggestion by these AECL researchers that *“the measurements are at fault”* is that the data in question are from passive tritium monitors which were found to be consistently *low* by a factor  $\sim 2$  compared to the predicted values. Nevertheless, a comparison of OPG passive monitors at Pickering and Darlington with the supposedly more accurate active monitors, *located at the same sites*, showed that the passive samplers gave tritium concentrations that were typically 2 times *higher* than their active counterparts. At the same time, experiments with passive monitors spiked with a known amount of tritium have shown that as much as 40% of the tritium was lost during summer testing periods.

These results leave us with the contradictory situation that while passive samplers are prone to *lose* previously adsorbed tritium, they can nonetheless collect *and retain* more tritium than active monitors at the same site. The only way this discrepancy can be resolved is to conclude that active tritium monitors are much less efficient at collecting and retaining tritium than is usually assumed, and may thus generate tritium concentration data that is as much as 4 times lower than the actual concentration.

The CNSC have also reported similar differences between measured and predicted tritium in air concentrations at the Bruce site, (See the CNSC’s 2009 Report No. INFO-0792). However, one can only wonder why it took the CNSC almost ten years to recognize these discrepancies; but it is even more disconcerting that our nuclear operators have taken no steps to address this problem and it raises the question of how reliable *any* of OPG’s tritium-in-air data really are – a topic that’s addressed in the next Section.

## **9.2 Tritium-in-Air Monitoring in the Vicinity of the Bruce Site**

It has already been noted that *passive* tritium-in-air monitoring at locations near Bruce NGS was found to be unreliable and was abandoned in 2011. This left *active* tritium monitoring, which had been on-going at ten designated sites, as the method used for monitoring airborne tritium in the environment near Bruce NGS. *Active* tritium monitors draw air at a known flow rate through stainless steel canisters containing Type 3A molecular sieve which absorbs water

vapor, including *tritiated water vapor*, from the ambient air. The molecular sieve canisters are replaced with fresh canisters every month and the absorbed tritiated water on the exposed molecular sieve is extracted and analyzed by liquid scintillation counting.

The Canadian Radiological Monitoring Network, (CRMN), also operates *active* tritium monitors using molecular sieve absorbents at their own monitoring installations near the Bruce site and have reported tritium in air data for the period 2004 to 2013 that are claimed to be “*consistent with results reported from other monitoring activities in close proximity to Canadian nuclear power stations*”. The CRMN further notes:

*The CNSC is responsible for regulating nuclear power facilities and maintains strict control of environmental radioactivity release from nuclear reactors, which is performed under rigorous monitoring conditions.*

It would be nice to believe these assurances that all is well with the air monitoring data that is generated by active tritium in air monitors but, as shown below, this is demonstrably untrue.

### **9.3 Active Tritium Monitoring Data**

An obvious concern with active tritium-in-air monitors is the use of a finite amount of molecular sieve absorbent because, after prolonged use, the absorbent must eventually become saturated with moisture and ceases to register the presence of any additional tritiated water vapor. To quantify this problem, we need to know the water absorbing capacity of Type 3A molecular sieve which is about 22 grams of H<sub>2</sub>O per 100 grams of absorbent at 25 °C and 75 % R.H. Now since air at 25 °C may contain up to 25 g of H<sub>2</sub>O per m<sup>3</sup> of air, 1 kg of molecular sieve may saturate after exposure to as little as 8.8 m<sup>3</sup> of air. It follows that at 25°C, and for an air flow rate of 1 m<sup>3</sup> per day, the molecular sieve in a tritium monitor is potentially fully saturated after only 8.8 days.

The performance of tritium monitors with regard to saturation of the absorbent was investigated in some detail in a U.S. study carried out in the period 2000 to 2001, as reported in: “*A Comparison of Desiccant Materials used for Monitoring Atmospheric Tritium Concentrations in a Semi-Arid Climate*”, INEEL Oversight Program Report, issued 2001. In this report the results are presented for an experimental study carried out at Idaho Falls which compared the effectiveness of molecular sieve and silica gel absorbents in collecting atmospheric moisture for the measurement of the concentration of tritium in air. The first point to note in this study is that the monitor sampled air at a flow rate of 1.9 liters/min, which equals 2.7 m<sup>3</sup>/day, and some cases of absorbent saturation were observed after less than 1 month of monitor operation. However, it is also noteworthy that this 2001 U.S. study recorded other problems with tritium monitoring utilizing molecular sieve absorbents:

*The atmospheric moisture collection efficiency for molecular sieve decreases as the ambient temperatures increase. As shown in other studies (4 references provided), a fraction of the moisture adsorbed was driven off by relatively warm dry air. This is evident in a decrease in the moisture collection efficiency observed for the molecular sieve at higher temperatures.*

The air temperature range of this U.S. study was 23 °C to –12 °C, i.e. a range that is comparable to the annual temperature range for Ontario.

Returning to our discussion of the use of active tritium monitors in the vicinity of Canadian nuclear facilities, the CRMN claim to have addressed the problem of absorbent saturation by using very low flow rates as described on the Health Canada website:

*The flow rate of air through the tritium monitor is reduced during the summer, from 0.2 to 0.07 m<sup>3</sup> per day, to prevent the sieve material from becoming saturated due to high humidity.*

Unfortunately, however, while the use of reduced flow rates may solve the problem of absorbent saturation, questions about the reliability of the resulting data remain. Thus, at the CRMN sampling flow rate of only 0.07 m<sup>3</sup> per day, we have an accumulated monthly volume of (30 × 0.07) m<sup>3</sup> or 2.1 m<sup>3</sup>. Now, let's compare this volume to the volume of air "sampled" by a human receptor at the same site who typically breathes at a rate of about 20 m<sup>3</sup> per day. In a month, this amounts to 600 m<sup>3</sup> or almost 300 times more air than a CRMN tritium monitor samples. There are a number of unavoidable consequences to this under-sampling of ambient air by the CRMN tritium monitors, including poor liquid scintillation counting statistics and an inability to detect short-term fluctuations in airborne tritium during periods of high winds.

However, regardless of these concerns, the ultimate test of the performance of a tritium monitor is how well the monitoring data correlates with site emissions data. To this end, we have used 2006 data from the B2, B3 and B4 monitoring sites, and compared the values to monthly data for tritium emissions for the entire Bruce site, namely the summed tritium emissions from Bruce A, Bruce B and the WWMF, as shown in Figure 1.

**Figure 1: Bruce Airborne Tritium Emissions and Monitoring Site Data for 2006**

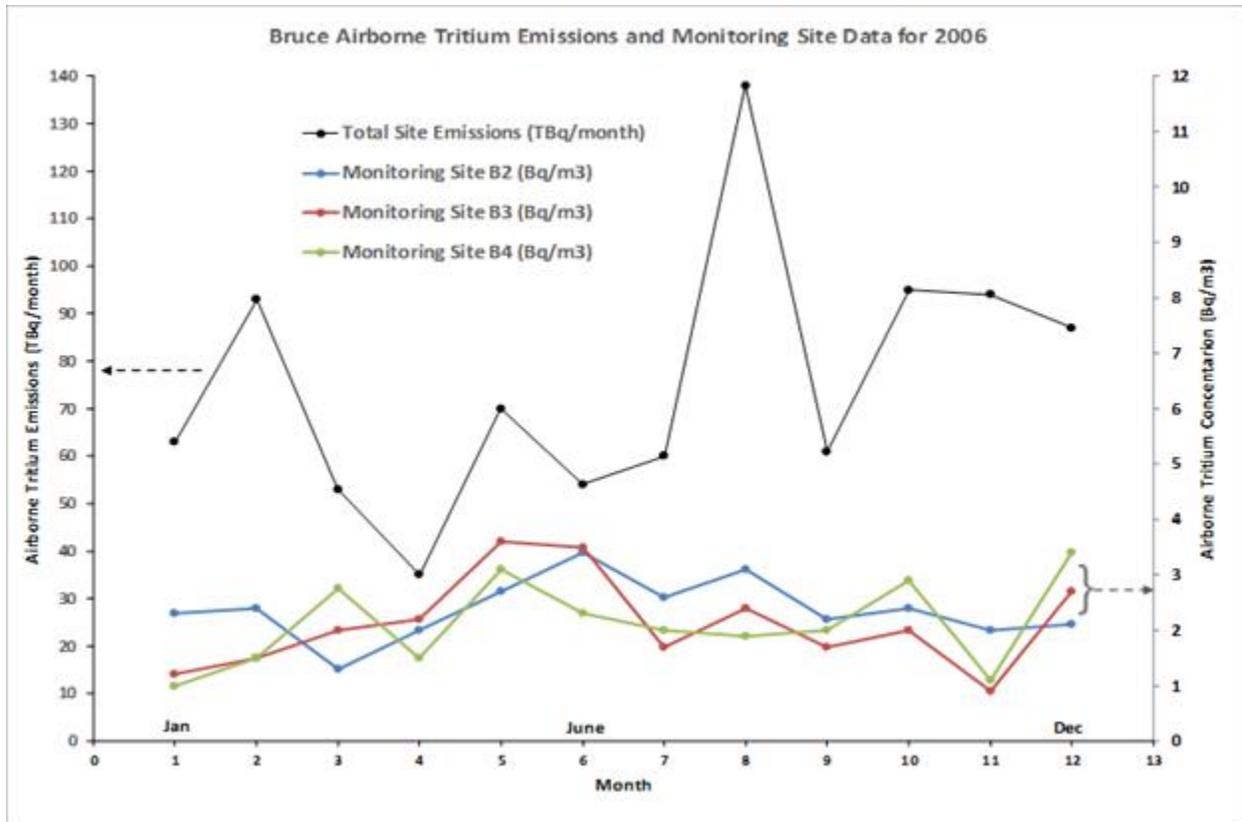
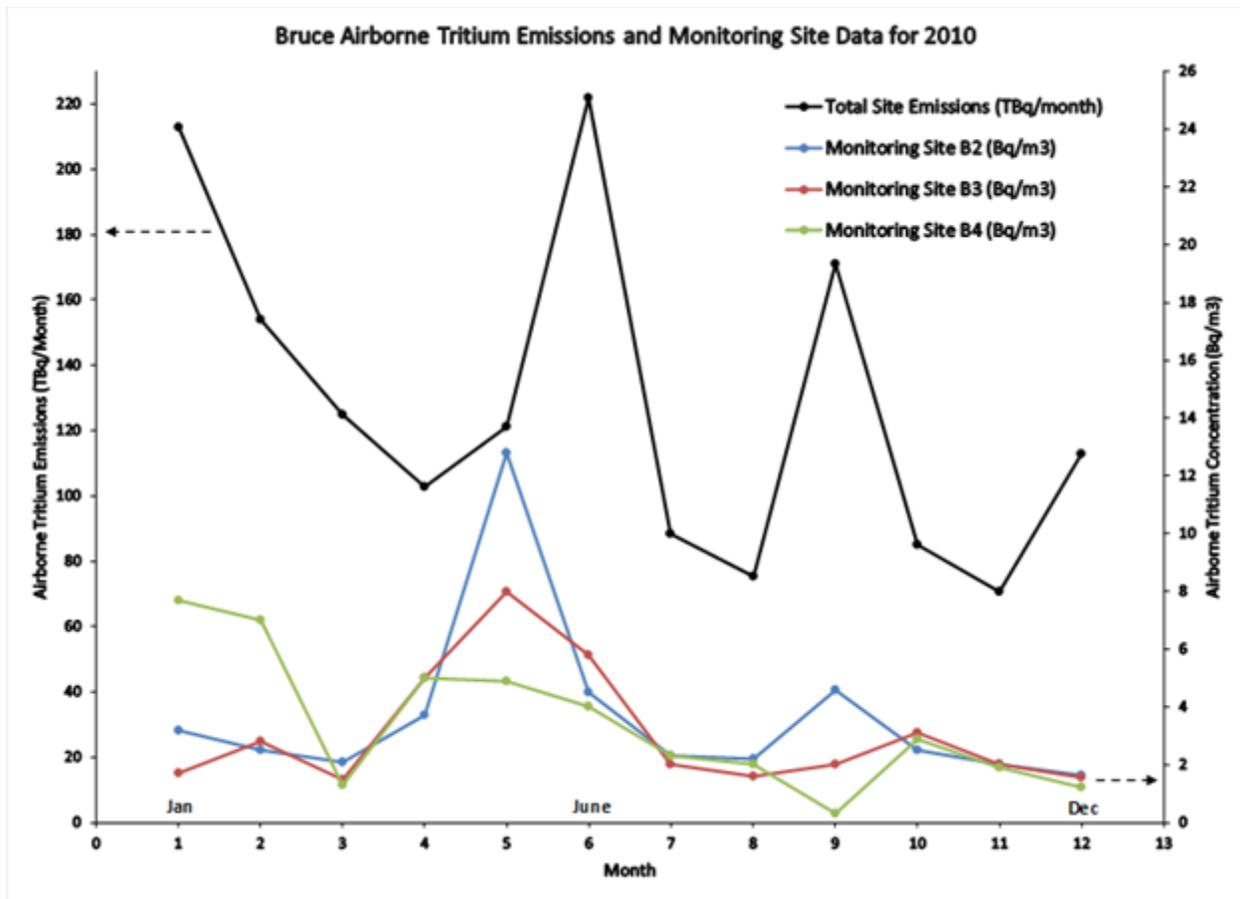


Figure 1 shows that in 2006 there were two periods, one in February and the other in August, when tritium emissions from the Bruce site peaked. There was also an extended period, from October to December, when tritium emissions were significantly above the average emission rate of 75 TBq/month. By comparison, the three off-site monitors, B2, B3 and B4, exhibited the highest tritium in air concentrations in the period May to June and were otherwise unresponsive to changes in station emissions.

Figure 2, below, shows the equivalent data plotted for 2010, and once again we see there is a poor correlation between the Bruce site tritium emissions and the B2, B3 and B4 monitor concentrations. However, in 2010 we also see one month (May) where these monitors recorded a tritium emission event that does not coincide with any of the major peaks in the site's emissions such as those that occurred in June and September.

**Figure 2: Bruce Airborne Tritium Emissions and Monitoring Site Data for 2010**



A possible explanation for the airborne tritium monitor peaks seen in May 2010, particularly at the B2 site, would be that there was a substantial tritium release from the WWMF incinerator at that time. We therefore need to estimate the maximum probable emission rate, (MPER), of tritium from the WWMF incinerator and the associated tritium concentration at the B2 monitoring site. The average specific activity of tritium in the boxes and bales of waste that are incinerated at the WWMF is  $2.55 \times 10^{11}$  Bq/m<sup>3</sup>, (See, for example, *Reference Low and Intermediate Waste Inventory for the Deep Geologic Repository*, Report No. 00216-REP-03902-00003-R0032010). If we assume that this waste was incinerated at a maximum rate of 1 m<sup>3</sup> per hour, we have a tritium MPER of  $2.55 \times 10^{11}$  Bq/hour which is equal to  $7.08 \times 10^7$  Bq/s.

The B2 monitoring site is located 1 km due east of the WWMF incinerator in which case, (See Section 9.1), the atmospheric dilution factor (X/Q) is  $1.58 \times 10^{-6}$  s/m<sup>3</sup> and the maximum airborne tritium concentration at the B2 site due to operation of the WWMF incinerator would be  $7.08 \times 10^7$  (Bq/s)  $\times$   $1.58 \times 10^{-6}$  (s/m<sup>3</sup>) or 112 Bq/m<sup>3</sup>. This is about 7 times *larger* than the 16 Bq/m<sup>3</sup> estimated from the 2010 annual tritium emissions from Bruce B – the tritium source which is the closest station to the B2 monitoring site. Thus, the notion that tritium emissions from the WWMF are minor compared to tritium emissions from either Bruce A or Bruce B is simply not valid.

One additional point that needs to be addressed with regard to tritium emissions from the WWMF, is the nature of the effluent gases from the incinerator stack. These gases have been “scrubbed” to reduce the concentrations of some of the more corrosive species, (e.g. NO<sub>2</sub>, SO<sub>2</sub> and HCl), to acceptable levels. During this process the effluent gases cool and become saturated with water vapor. Additional cooling of these gases occurs due to contact with the surfaces of the incinerator ductwork, causing water droplets to form in the gas stream. Upon release from the stack, these water droplets rapidly evaporate, withdrawing the latent heat of vaporization from the ambient air and cooling the plume. The resulting negative buoyancy of the emergent gases – that is a plume with a higher density than the ambient air – results in fumigation of the ground in the vicinity of the stack.

The preceding discussion shows that the atmospheric dispersion of incinerator off-gases is quite different to and more locally concentrated than the atmospheric dispersion of the much cooler and dryer gaseous emissions from a nuclear plant. In the case of the Bruce nuclear site this means that the WWMF incinerator emissions are likely to be more detrimental to outside workers and members of the public living within a few kilometers of the Bruce site, than emissions from Bruce A and Bruce B because the latter are more diluted from being more widely dispersed.

Another issue concerning the nature of tritium emissions from the WWMF that has only been briefly mentioned in this report, (See Section 9.0), is the fact that the waste incinerator emissions are “spiked” because they are intermittent by nature. However, DRLs are calculated assuming emissions are *continuous* and deviate little from an average value over a one-year interval. Thus, it is assumed (without proof) that accurate doses to members of a critical group may be derived from long-term emissions data. However, as demonstrated in detail below, such doses are significantly *underestimated* if intermittent releases are simply added together and averaged as if they were continuous releases.

Furthermore, the chance coupling of an emission spike with a period of heavy precipitation serves to amplify the dose to a critical group, particularly if they are farmers using well water. To quantify these dose enhancement effects, it is necessary to compare doses calculated for short-term acute releases with doses derived from annual average “long-term” releases.

For off-site doses from short-term releases it is normally assumed that the wind is blowing steadily in the direction of a receptor under stable atmospheric conditions. Such conditions assume no crosswinds or plume meander and are also assumed to

prevail for periods of up to 2 hours following a spiked or intermittent release. The receptor is thus exposed to the maximum possible concentration of the released radionuclide. The associated atmospheric dilution factor is usually referred to as the “2-hr (X/Q)” where X is the atmospheric concentration of the species of interest, in units of Bq/m<sup>3</sup>, and Q is its release rate from the source in units of Bq/s.

By comparison, for the assessment of long-term releases, a sector-averaged Gaussian plume model is usually adopted rather than one in which the plume is directed at the receptor location throughout the exposure period. For this reason, long-term release calculations typically use site-specific meteorological data to construct joint-frequency tables which categorize hourly observations of wind speed and direction into 7 stability classes, 16 compass directions, and 6 wind-speed ranges. Such frequency tables are then used to calculate annual-average atmospheric dilution factors, also referred to as long-term (X/Q)s.

An illustration of the magnitude of the difference between short-term and long-term (X/Q)s may be found in the Bruce Power Report: “*Updated Site Specific Atmospheric Dilution Factors for Use in Safety Analysis*” Report No. B-Rep-03611-00001, issued in June 2004. Here we find “*Recommended Atmospheric Dilution Factors or (X/Q)s for the Bruce Site*” as follows:

$$\text{Short-Term ADF} = 86 \times 10^{-6} \text{ s/m}^3$$

$$\text{Long-Term ADF} = 1.7 \times 10^{-6} \text{ s/m}^3$$

Thus we see that doses from short-term releases are about 50 times higher than doses from the same total releases delivered continuously over a long-term interval, e.g. 1 year. However, as noted above, an intermittent release of tritium may also be accompanied by periods of heavy precipitation leading to so-called washout and wet deposition of tritium and even higher doses.

Tritium washout near nuclear facilities has been discussed in the open literature – see for example: “*Investigation of the Environmental Fate of Tritium in the Atmosphere*” CNSC Report INFO-0792, issued 2009, but the full extent of this phenomenon is very complex and difficult to model. Nevertheless, for the purposes of the present study a simplified formalism may be used as described below:

The wet deposition flux  $F_w$  (Bq.m<sup>-2</sup>.s<sup>-1</sup>) is assumed to be proportional to the tritium release rate multiplied by a washout coefficient  $\Lambda$  which is the fraction of HTO removed per second. Allowing for plume dispersion we then have,

$$F_w = Q \Lambda \Phi / (x \bar{u} \theta)$$

Where,

Q is the tritium release rate in  $\text{Bq}\cdot\text{s}^{-1}$

$\Lambda$  is the washout coefficient in  $\text{s}^{-1}$

$\Phi$  is the joint frequency of occurrence of wind direction and rainfall in the receptor sector

$x$  is the distance from source to receptor in m

$\bar{u}$  is the average wind speed in  $\text{m}\cdot\text{s}^{-1}$

$\theta$  is the angular width of the sector in radians

It follows that the concentration of tritium in rainwater,  $C_w(\text{Bq/liter})$ , is given by:

$$C_w(\text{Bq/liter}) = F_w T / (1000 P)$$

Where,

T is the duration of the study period in s

P is the total amount of rain in the study period in m

1000 is the factor to convert  $\text{Bq}\cdot\text{m}^{-3}$  to  $\text{Bq/liter}$

This washout coefficient formalism has been used to calculate the concentration of tritium in rainwater for realistic examples of annual average and short-term releases, namely 200 Ci/week vs. a short-term release of 1700 Ci. The average wind speed was set at  $3 \text{ m}\cdot\text{s}^{-1}$  and the washout coefficient was conservatively assumed to be  $1 \times 10^{-5} \text{ s}^{-1}$ . The annual average rainfall at the WWMF site is about 1000 mm and a short-term (24 hr) rainfall event of 15 mm (accompanied by a tritium emission spike) was assumed. The resulting tritium concentration in rainwater was calculated to be about 20,000  $\text{Bq/liter}$  for a short-term release – a value that is about 100 times *higher* than the concentration calculated from the average annual release.

Thus, to summarize: Using long-term averaging of intermittent tritium emissions leads to a significant *under-estimation* of doses to exposed individuals. This effect comes about because tritium concentrations downwind from a source decrease with time due to increased plume meander. Thus, the long-term calculation averages out the dose impact to a specific group by sharing it with a hypothetical “average dose recipient”. However, real emission peaks are capable of delivering much larger doses to real individuals located at the plume’s point of impingement; furthermore, the dose is only *increased* by concurrent periods of precipitation.

## 9.4 Carbon-14 Emissions

### A. C-14 Doses to the Public

The total internal dose from C-14 emissions is made up of an ingestion component and an inhalation component. It is acknowledged that the ingestion component may make a significant contribution to the dose from exposure to C-14, but for the present discussion we wish to focus on the contribution from inhalation. The inhalation dose previously expressed by Eqn. 1 for the case of tritium may be modified for C-14 as follows:

$$\text{Inhalation Dose (Sv/yr)} = F_{C-14} \times BR(\text{m}^3/\text{s}) \times (X/Q) (\text{s}/\text{m}^3) \dots \text{Eqn. 4}$$

Where,

$$F_{C-14} = R_I \times DCF_I (\text{Sv}/\text{Bq}) + R_O \times DCF_O (\text{Sv}/\text{Bq}) + R_P \times DCF_P (\text{Sv}/\text{Bq})$$

With,

$$R_I = \text{Inorganic C-14 Release (Bq/yr)}$$

$$R_O = \text{Organic C-14 Release (Bq/yr)}$$

$$R_P = \text{Particulate C-14 Release (Bq/yr)}$$

and,

$DCF_I$  is the dose conversion factor for inorganic C-14

$DCF_O$  is the dose conversion factor for organic C-14

$DCF_P$  is the dose conversion factor for particulate C-14

The reason for making a distinction between inorganic, organic and particulate C-14 is that the dose conversion factors for these species are quite different as noted by P. Humphreys et al. in *The Speciation of H-3, C-14 and Cl-36 in the DGR System* Report No. QRS-1335B-TR4, issued in December 2011.

For this study, we have taken dose conversion factors from Table 3.4 of Humphrey et al's report and have assumed that for WWMF emissions the inorganic carbon is  $^{14}\text{CO}_2$ ; the organic carbon is  $^{14}\text{CH}_4$  and the particulate  $^{14}\text{C}$  is an S-type aerosol. Hence:

$$DCF_I = 6.2 \times 10^{-6} \mu\text{Sv}/\text{Bq}$$

$$DCF_O = 2.9 \times 10^{-6} \mu\text{Sv}/\text{Bq}$$

$$DCF_P = 5.8 \times 10^{-3} \mu\text{Sv/Bq}$$

For the WWMF, C-14 emissions have contributions from all three of these species because of the multiple sources of C-14 at this site as summarized in Table 5, below. With the exception of <sup>14</sup>C-particulate, the breakdown of species released by each source listed in Table 5 was derived from the previously noted report by P. Humphreys et al.: *The Speciation of H-3, C-14 and Cl-36 in the DGR System*.

For <sup>14</sup>C-particulate it is known that the WWMF incinerator emits ~ 1 mg of particulate per m<sup>3</sup> of exhaust gases containing about  $1 \times 10^7$  Bq of C-14 per kg of incinerated waste, and this enters the environment at a volumetric flow rate of 1.7 m<sup>3</sup>/s. If it is assumed that the incinerator operates 25 % of the time and the C-14 specific activity of the waste is  $1 \times 10^7$  Bq/kg, it may be estimated that <sup>14</sup>C-particulate contributes about 4% to the incinerator C-14 emissions, as shown in Table 7.

**Table 7: Carbon-14 Speciation in Airborne Emissions from the WWMF**

Sources of <sup>14</sup> C	<sup>14</sup> C Speciation
Degassing of compacted bales and boxes	95% <sup>14</sup> CO <sub>2</sub> + 5% <sup>14</sup> CH <sub>4</sub>
Degassing of spent moderator resin	90% <sup>14</sup> CO <sub>2</sub> + 10% <sup>14</sup> CH <sub>4</sub>
Incinerator off-gasses	96% <sup>14</sup> CO <sub>2</sub> + 4% <sup>14</sup> C-particulate

The annual C-14 inhalation doses at BF14 due to airborne releases from the WWMF have been calculated using Eqn. 4 with the following input parameters:

- (i) The annual *monitored* C-14 release data in the OPG's report: *Western Waste Management Facility Application for Licence Renewal*
- (ii) The annual *unmonitored* C-14 releases reported in Section 4.0 of this report. These are  $1.53 \times 10^{11}$  Bq/year for LLW and  $3.55 \times 10^{11}$  for ILW, (i.e. moderator resin)
- (iii) These annual releases were divided into inorganic, organic and particulate <sup>14</sup>C components using the ratios reported in Table 5
- (iv) The breathing rate was set at 1.5 m<sup>3</sup>/hr

(v) The (X/Q) for the BF14 site, as previously determined, was  $5.617 \times 10^{-7} \text{ s/m}^3$ . With these input parameters, and for the period 2007 to 2012, the total annual C-14 inhalation doses, (i.e. doses due to the sum of the annual contributions from monitored and unmonitored emissions), are estimated to have been about 50 % higher than the doses estimated by OPG.

### **B. C-14 Doses to WWMF Workers**

Doses to WWMF workers are dominated by the contribution from inhalation of airborne C-14. For work at the WWMF site carried out *outside* the waste storage buildings, Eqn. 1 of this report has been used to calculate a conservative inhalation dose. For the calculation, the following input parameters were used:

- (i) A worker's site occupancy factor of 0.1
- (ii) An annual release of C-14 from monitored and unmonitored sources at the WWMF site of  $5 \times 10^{11} \text{ Bq}$ , as reported in OPG's *Preclosure Safety Assessment*
- (iii) A dose conversion factor of  $6.2 \times 10^{-6} \text{ } \mu\text{Sv/Bq}$
- (iv) A breathing rate of  $1.5 \text{ m}^3$  per hour
- (v) A dilution factor (X/Q) of  $1.58 \times 10^{-5} \text{ (s/m}^3\text{)}$  for an average 100-meter distance to WWMF C-14 emission sources

We have also considered the dose to a WWMF worker who spends an average of 1000 hours per year *inside* a LLSB and works with a breathing rate of  $1.5 \text{ m}^3/\text{hour}$ . OPG's *Preclosure Safety Assessment* Report states that the concentration of C-14 inside a LLSB is  $3200 \text{ Bq/m}^3$ . The dose to a WWMF worker for each of these scenarios is then calculated to be as follows:

Dose to an outside worker =  $64.4 \text{ mSv/year}$

Dose to an inside worker =  $0.03 \text{ mSv/year}$

Evidently, the C-14 dose estimated for an outside worker is much higher than the dose estimated by OPG and exceeds the annual regulatory limit of  $50 \text{ mSv}$  by a significant margin.

## **10.0 Conclusions and Additional Discussion**

The most important safety requirement of a nuclear waste disposal site is that the owner/operator has reliable, scientifically based data on the site's radionuclide inventory and a detailed accounting of all, (chronic and acute), airborne and waterborne radionuclide releases to

the environment. Most unfortunately, for OPG's Western Waste Management Facility, these basic safety requirements are *not* being met for two reasons:

- (i) OPG's inability to reliably estimate the WWMF radionuclide inventory
- (ii) OPG's failure to include in its *Western Waste Management Facility Application for Licence Renewal* report, data on fugitive or unmonitored emissions from this site.

OPG and the CNSC have been subject to criticism over these issues in the past, especially at the Deep Geological Repository (DGR) Hearings held in the period 2012 to 2014. Remarkably, and incorrectly, the CNSC concluded that even if there was some basis for these criticisms, *the safety case for the proposed DGR was not affected*. This claim by the CNSC is simply not true as explained in item (i) below:

**(i) The WWMF and DGR Radionuclide Inventory**

In 2010 OPG published a report entitled: *Reference Low and Intermediate Waste Inventory for the Deep Geologic Repository*, Report No. 00216-REP-03902-00003-R0032010. This report forms the basis of the environmental safety assessment of the proposed DGR and was mostly derived from data on the radioactive waste already in storage at OPG's WWMF. Thus, any questions concerning the accuracy and validity of the data in OPG's 2010 DGR *Waste Inventory* report, are by default also questioning the radionuclide inventory of the WWMF.

On March 21<sup>st</sup>, 2014, the CNSC were requested by the Deep Geologic Repository Joint Review Panel, (DGR JRP), to prepare a response to information request EIS 13-514. This request was dispositioned by the CNSC in June 2014, as described in CNSC File:2.05 (e-Doc: 4441634). Here we find the *CNSC Staff Sufficiency Review for IP #13-EIS-13-514* which included the following statement on page 5:

*“Dr. Frank Greening raised questions on the accuracy of OPG's 2010 Reference Waste Inventory of low and intermediate level waste that would be emplaced in the proposed DGR. These questions concern concentrations in CANDU pressure tubes and garter springs for which the concentrations of some radioisotopes appear to have been significantly underestimated or not estimated at all by OPG.*

*OPG prepared revised radionuclide concentration data for the retube (pressure tube) packages, including garter springs. For normal operations, the revised pressure tube concentrations reflect the changes in the inventories decayed for 10 years. This results in a package dose rate that is higher than the DGR Waste Acceptance Criteria (WAC) of 2 mSv per*

*hour; whereas the original modelled retube (pressure tube) waste package would be in compliance with the DGR waste acceptance criteria. Packages greater than the DGR waste acceptance criteria would not be accepted at the DGR without further shielding or decay. However, for conservatism and as a direct comparison, OPG assumed the package containing the revised inventory of retube waste was accepted as-is at the DGR for their revised radiological dose assessments.*

*OPG conducted the radiological dose assessments with the revised pressure tube inventory re-evaluating Scenario 2 from the DGR Preliminary Safety Report [12] that examined the impact of the pressure tube waste packages in the loading areas within the Waste Package Receipt Building (WPRB).....*

*The revised retube package would be the largest contributor to the doses to workers. The potential external dose rates to the workers (same receptors presented in the Preliminary Safety Report [12]) increased by approximately four (4) times. This is attributed mainly to the increase in the Cobalt-60 concentration.*

*OPG stated that since the DGR waste acceptance criteria would not be met by the revised retube waste package as-is, additional shielding or a longer decay time would be required before this waste could be transferred to the DGR, thereby decreasing the external dose rate to the workers to accepted levels.... Thus, it is recognized that there is an increase in the external dose rates to DGR workers due to the revised pressure tube inventories.*

So we see that, in spite of CNSC claims to the contrary, OPG acknowledges that the safety case for the DGR was affected in a negative way by errors that I identified in its 2010 Reference Waste Inventory of radioactive waste slated for emplaced in the proposed DGR – and a large percentage of this waste is already stored in OPG’s WWMF. Unfortunately, however, there are many remaining errors and inconsistencies in the current estimates of the WWMF radionuclide inventory that need to be addressed.

**(ii) Fugitive Emissions from the WWMF**

OPG does acknowledge the existence of unmonitored emissions from the Bruce site when, for example, it states on page 38 of its 2009 *Preclosure Safety Assessment* report, (in reference to Table 3.28 which lists the 2005 to 2007 H-3 and C-14 airborne emissions from the WWMF):

*The Bruce Site emission rates include the station stacks, the WWMF and other monitored points, but not any fugitive emissions\**

\* My emphasis

However, OPG consistently excludes “*fugitive emissions*” from any tabulations of the annual radiological emissions from the WWMF, and also excludes, without justification, any contributions from fugitive emissions in its dose calculations for the WWMF – a practice apparently condoned by the CNSC. Thus, at the 2002 Licencing Hearing for the WWMF, the CNSC had this to say about fugitive emissions:

*Although the WWMF is designed to prevent and contain uncontrolled dispersion of radioactive or other hazardous substances, some fugitive emissions of volatile tritium and C-14 escape, mostly from waste storage. CNSN staff expressed the view that these releases are small and do not pose a significant risk to the workers or the public*

Nevertheless, in this report, we have evaluated the doses due to tritium and carbon-14 fugitive emissions from the WWMF and shown that they are frequently comparable to, or in the case of C-14, much larger than doses attributable to monitored emissions. Indeed, OPG has recently acknowledged in its response to the DGR EIS-01-08, that the primary waste package in the WWMF is non-processible bins and drums which collectively have the largest overall inventory of tritium and carbon-14. In addition, OPG admits that these waste containers are “*likely the main source of tritium emissions from the WWMF*”. I would therefore say to the CNSC: Until these un-monitored emissions are quantified and included in dose estimates, OPG should be denied a Licence for the continued operation of the WWMF.

### **(iii) Why is OPG Incinerating *any* Radioactive Waste at the WWMF?**

In Section 3.0 of this report, we noted that over the past 5 years ~ 20 % of the LLW received by the WWMF was incinerated rather than stored. This fact begs the question: Why does OPG incinerate only 20 % of the waste received every year at the WWMF; or conversely, why does OPG need to incinerate *any* radioactive waste? There appears to be one simple answer to this question: Economics, because it’s a lot cheaper to incinerate waste than to store it.

The cost differential between incinerating vs. storing LLW is spelled out in detail in OPG’s response to an AMPCO interrogatory at the 2014 Ontario Energy Board Hearings on OPG’s 2014/2015 Payment Amounts Application – See Filing EB-2103-0321, Exhibit L, dated March 19<sup>th</sup>, 2014. Here we find that the annual cost of storing LLW is \$1,062/m<sup>3</sup>, compared to a disposal (by incineration) cost of \$364/m<sup>3</sup>. From this difference in cost, and allowing for the fact that the ash residue from the incineration of waste must be stored, we are able to calculate annual cost savings to OPG from incinerating, instead of storing LLW, as shown in Table 8.

**Table 8: Annual Cost Savings from Incinerating vs. Storing LLW at the WWMF:  
Data for the period 2007 to 2015**

Year	Incinerated Volume (m <sup>3</sup> )	Annual Cost Savings (\$ Cdn.)
2007	966	646,541
2008	1196	800,480
2009	874	584,966
2010	1332	882,504
2011	1437	961,780
2012	530	354,728
2013	600	401,578
2014	397	265,711
2015	499	333,979

However, it should be recognized that the savings for a particular year may be carried over to all subsequent years since the money saved in one year by *not* storing a shipment of waste is saved *in perpetuity* by incinerating the waste. Thus, we find for the period in question – 2007 to 2015 – the total savings to OPG from incinerating, rather than storing, LLW is about \$32 million.

Nevertheless, by incinerating an average of 870 m<sup>3</sup> of LLW per year over this 9-year period, OPG has knowingly and without any justification released substantial quantities of tritium and carbon-14 to the environment, as listed in Table 9, as well as significant amounts of I-131, beta-particulate and un-monitored quantities of alpha-particulate – releases that could have been largely avoided if the LLW in question was simply stored in LLSB's at the WWMF instead of being incinerated; but this raises serious questions with regard to OPG's adherence to Canadian Radiation Protection Regulations, as further discussed in item (v) below.

**Table 9: WWMF Incinerator Annual Airborne Emissions of Tritium and Carbon-14:  
Data for the period 2007 – 2015**

Year of Operation	WWMF Annual Airborne Emissions	
	Tritium (Ci)	Carbon-14 (Ci)
2007	362	0.126
2008	735	0.130
2009	1338	0.106
2010	784	0.162
2011	538	0.108
2012	281	0.051
2013	386	0.053
2014	194	0.042
2015	112	0.038

**(iv) Why does OPG want to replace the WWMF with a DGR?**

In its *License Renewal Application Report* for the WWMF OPG states:

*WWMF has been operating safely since it was established in 1974. The additional buildings and structures would not alter the basic purpose and activities associated with the WWMF. The ongoing operation of WWMF will enable the nuclear generating stations in Ontario to continue operating as planned under their current respective operating licenses.*

*This report presents information on the performance of WWMF in areas related to the fourteen Safety and Control Areas. During the current licensing period, WWMF has operated safely and reliably to protect the public, the workers and the environment. OPG is proud of its excellent record in conventional and radiological worker safety, and is well positioned for the continued operation of WWMF.*

Given this glowing assessment of the WWMF’s performance over 40+ years of operation, one has to wonder why OPG would want to move the waste currently stored at a “safe” facility – one that “enables the nuclear generating stations in Ontario to continue operating as planned” – and transfer all of this waste to a DGR. Furthermore, the fact that OPG wants to start moving waste from the WWMF to a DGR as soon as possible, suggests that there are problems with

the continued storage of waste at the WWMF. So, what could these problems be that only a DGR can fix? However, before answering this question, let's first look at the arguments that have been used to justify the construction of a DGR:

- (i) The WWMF is running out of storage space
- (ii) It's more cost effective to store L&ILW in a DGR than in the WWMF

Argument (i) is certainly not valid because OPG has been continuously adding more storage space at the WWMF since its inception in the 1970s. Furthermore, OPG has indicated in its current *Western Waste Management Facility Application for Licence Renewal* that it intends to build six additional LLSBs at the site as follows: LLSB 15 and 16 in 2019; LLSB 17 in 2023; LLSB 18 in 2025; LLSB 19 in 2028; LLSB 20 in 2031. These new storage buildings would provide approximately 40,000 m<sup>3</sup> of new storage space for L&ILW. In addition, OPG has stated that it makes decisions on when to construct new buildings *approximately 5 years before they are required* to ensure sufficient time in advance of the use of the existing available storage space, to allow for the design, site preparation and construction activities.

Argument (ii), concerning the cost of waste storage and disposal, is also highly questionable when the cost of constructing and operating a DGR is compared to the operating costs of the WWMF. Using data reported by OPG to the Ontario Energy Board we find that the cost of constructing a DGR, retrieving L&ILW from the WWMF and transferring it to the DGR would be \$6250/m<sup>3</sup>. By comparison, the cost of storing the existing and future (anticipated) wastes at the WWMF, including the construction of new LLSBs as required, would be \$1510/m<sup>3</sup>, or about 4 times *less* than the DGR option.

A clue to OPG's only *legitimate* reason for moving waste from the WWMF to a DGR may be found in an OPG letter by K.E. Nash to B. Howden at the CNSC entitled: "*Deep Geologic Repository for Low and Intermediate Level Waste*", OPG Report No.CD# W-CORR00531-00171, issued March 7, 2005. In Section 1.3 of this letter OPG claims that a DGR is needed because "*it provides a greater margin of safety than the existing facilities*". However, in contradiction to this assertion, the *Environmental Risk Assessment for the Western Waste Management Facility*, presented in OPG Report: 01098-REP-07701-00011-R000, issued April 2016, states:

- (i) *The operation of the WWMF presents no radiological risk to the public*
- (ii) *Non-radiological emissions resulting from the operations at the WWMF are compliant with the standards protective of human health and therefore no human health effects are likely.*

- (iii) *The risk evaluation for ecological receptors shows there are no adverse effects due to exposure to radiological contaminants.*
- (iv) *Radiological contaminants do not pose an adverse effect to biota at the WWMF. No further evaluation with respect to radiological contaminants is recommended.*

Here we see OPG engaging in “double-speak” because on the one hand it claims that the existing WWMF may be relied upon to protect the health and safety of the public and the environment for years to come, while on the other hand, OPG claims that a DGR is a better place to dispose of our nuclear waste because “*it provides a greater margin of safety than the existing facility*” – namely, the WWMF. But if we look closely at OPG’s 2016 *Environmental Risk Assessment for the Western Waste Management Facility* report we find all sorts of inadequacies such as:

- (i) No mention whatsoever of *fugitive* emissions
- (ii) Ridiculous, self-fulfilling statements about worker safety

A good example of point (ii) would be:

*As it is expected that the health and safety of on-site workers is protected with the implementation of OPG’s Radiation Protection Program and Conventional Safety Program, no further risk assessment will be performed for on-site workers.*

Bruce Power claimed to have a robust *Radiation Protection Program* in place for the 2009 refurbishment of its Unit 1 & 2 reactors. Nevertheless, their program could not protect 557 workers from unmonitored exposures to airborne alpha-contamination. Evidently, *claiming* that something is safe falls far short of *proving* that it is safe!

**(v) Is OPG in Violation of Radiation Protection Regulations at the WWMF?**

In the latest edition of Canada’s *Radiation Protection Regulations*, we find a Section 4 entitled “*Radiation Protection Program*”, which reads in part as follows:

**4.** *Every licensee shall implement a radiation protection program and shall, as part of that program,*

- *(a) keep ... the effective dose and equivalent dose received by and committed to persons as low as is reasonably achievable\* , ..., through the implementation of*
- 

*(i) management control over work practices,*

- (ii) *personnel qualification and training,*
- (iii) *control of occupational and public exposure to radiation*

\*My Emphasis

The phrase “*as low as reasonably achievable*” is the well-known “ALARA” principle that has been adopted by the nuclear industry worldwide. The intent of this principle is to ensure that nuclear facility operators endeavor to *minimize* the exposure of workers and the general public to radiation, and in particular to *unnecessary or avoidable exposures*. And the CNSC appear to be a strong proponent of this concept as the following example shows:

On November 18<sup>th</sup>, 2016, the CNSC issued the following “Notice of Violation” to the Construction Company R.M. Belanger Ltd:

### **Violation**

*Failure to keep exposure to radiation to persons as low as reasonably achievable through the implementation of management control over work practices in violation of subparagraph 4 (a) (i) of the Radiation Protection Regulations. The extent and repeated nature of the non-compliances identified during the inspections conducted in August 2016 indicate that R. M. Belanger Limited does not have sufficient management control over work practices or the radiation protection program to keep doses as low as reasonably achievable.*

### **Administrative Money Penalty:**

AMP No. 2016-AMP-07: Amount of Penalty: \$ 4,900

In this particular instance, the CNSC shows that it is quite capable of fining a company with a radiation protection program that does not adhere to the ALARA principle. But the CNSC is apparently also willing to accept OPG’s practice of incinerating LLW at the WWMF, even though this unwarranted and totally avoidable method of dealing with radioactive waste is clearly in violation of the ALARA principle. Of course, OPG claims otherwise as in its *2015 Interim Status Report on Darlington, Pickering and Western Waste Management Facilities*, OPG report No. W-REP-00531-00009-R000, issued May, 2015. Here we read:

*OPG’s Nuclear Waste Facilities are designed to operate within regulatory limits and to ensure that radiological exposure to workers and the public, and impact on the environment are As Low As Reasonably Achievable (ALARA).*

However, the inconvenient truth is that OPG is in violation of subparagraph 4 (a) (i) Section 4 of Canada's Radiation Protection Regulations:

*To keep exposure to radiation of workers and the general public as low as reasonably achievable through the implementation of management control over work practices.*

Therefore, my recommendations with regard to the continued operation of the WWMF are:

- (i) Any future requests for a license for continued operation of the WWMF should be denied unless all emissions – *monitored and fugitive* – are quantified and substantially reduced
- (ii) An appropriate Administrative Money Penalty should be imposed on OPG for past violations of subparagraph 4 (a) (i) of Canada's Radiation Protection Regulations
- (iii) OPG should be ordered to stop incinerating radioactive waste at the WWMF facility

  
Hamilton, ON

March 15<sup>th</sup>, 2021