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Human health risks from mercury exposure from broken compact fluorescent lamps (CFLs)

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A B S T R A C T

Human health risk to infants/toddlers and adults was evaluated based on two exposure scenarios from compact fluorescent lamp (CFL) breakage; first in a room with no ventilation and no clean-up, and second in a room with adequate ventilation and clean-up. Concentration data from multiple exposure scenarios tested in a study by Stahler et al. (2008) were compared to human toxicity benchmarks to calculate hazard quotients. For the no clean-up scenario, hazard quotients were generally less than 1, suggesting an unlikely health risk. When the room was ventilated and the broken CFL was cleaned-up, mercury concentrations were generally lower. A review of release scenarios, along with duration-adjusted toxicity benchmarks, indicated that few releases produced levels of concern, but some scenarios resulted in exceedance of risk targets and require further study. Uncertainties in this screening characterization include assumptions about room size, ventilation, age of lamp, the distribution of mercury in the room, and also the choice of the toxicity benchmarks used to develop the hazard quotients.

1. Introduction

Compact fluorescent lamps (CFLs) are promoted as being more energy efficient and an eco-friendly replacement for incandescent lamps. Fluorescent lamps, including fluorescent tubes and CFLs, are increasingly being used in homes around the world as part of a drive to improve energy efficiency. Their use is promoted as a replacement option for incandescent bulbs by many federal and local government agencies. This screening evaluation of the potential human health risks from broken CFLs was conducted for the New Zealand Ministry for the Environment.

The key advantages of installing CFLs compared with incandescent lamps are large reductions in energy use and greenhouse gas emissions if the electricity is produced from burning fossil fuels (Parsons, 2006). A disadvantage of fluorescent lamps is that they contain milligram (mg) quantities of mercury. Mercury is an integral component of fluorescent lamps and a substitute chemical has not yet been identified. Internationally, concerns have been raised regarding potential mercury exposures following lamp breakage (Groth, 2008; Stahler et al., 2008).

1.1. Fluorescent lamps

Fluorescent lamps are electrical discharge lamps that contain low-pressure mercury vapor and an inert gas, usually argon. The inside of the glass is coated with a fluorescent phosphor powder. The mercury vapor is excited by an electrical current between two electrodes and emits ultraviolet (UV) light. The UV light causes the phosphor coating to fluoresce and emit visible light. Mercury (elemental or mercuric oxide) can be added to lamps in a variety of forms including liquid, solid, or pellet amalgam dosing technology (Parsons, 2006). A variety of mercury amalgams have been used in fluorescent lamps with varying combinations of iron, bismuth, indium, tin and lead (Parsons, 2006). During lamp use, the elemental mercury is oxidized and is adsorbed onto the phosphor powder, as well as onto other lamp components including the glass (Aucott et al., 2003; Jang et al., 2005; NJ MTF, 2002; Raposo et al., 2004; UNEP, 2005). Elemental mercury also becomes dispersed throughout the lamp during operations. These processes reduce the amount of mercury that can be volatilized (Aucott et al., 2003; NJ MTF, 2002) and so manufacturers add sufficient mercury to ensure that there is an adequate supply of mercury vapor present for the life of the lamp (Raposo et al., 2004; UNEP, 2005). A range of fluorescent lamps are available, but the focus of this analysis was on CFLs only. No data on the specific form of mercury in fluorescent lamps are available.

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Table 1
Amounts of mercury present in fluorescent lamps (mg per lamp) manufactured in various countries.

Country	Lamp type and amount of mercury per lamp (mg)	Reference
Europe	Halophosphate lamps 10	ROHs (2008)
Europe	Triphosphate lamps 5–8	ROHs (2008)
Canada	Linear fluorescent tubes 3–50	Environment Canada (2004)
United States of America	Linear fluorescent tubes 0–100	NEWMOA (2008)
United States of America	Linear fluorescent tubes 1.4–50	Culver (2008)
United States of America	Linear fluorescent tubes 1.25–5.96	Singhvi et al. (2008)
Australia	CFL 0.1–13	Boughey and Webb (2008)
Canada	CFL 1–25	Environment Canada (2004)
United Kingdom	CFL < 10	AEA Technology (2004)
United States of America	CFL 1–6	Culver (2008)
United States of America	CFL average 4	Energy Star (2008)
United States of America	CFL 5–50	NEWMOA (2008)

The quantity of mercury present in a fluorescent lamp depends on the type (linear versus CFL), brand, and the wattage (Aucott et al., 2003; Culver, 2008; NEWMOA, 2008; NJ MTF, 2002; Stahler et al., 2008). The mercury content is not listed on the packaging for many of the products sold and is not always easily accessible from manufacturer's websites. The amount of mercury reported is up to 30 mg per light lamp for CFLs and up to 115 mg for linear fluorescent tubes (Groth, 2008; Jang et al., 2005). Available international data on the mercury content of fluorescent lamps are summarized in Table 1. The amount of mercury per CFL can vary between brands as well as between lamps of the same type (Stahler et al., 2008).

Internationally, manufacturers are reducing the amount of mercury used in fluorescent lamps (Energy Star, 2008; UNEP, 2005). In 2007 the National Electrical Manufacturing Association (NEMA) introduced a voluntary cap on mercury content in lamps sold in United States (US) to 5 mg for CFLs less than 25 W and 6 mg for 25–40 W CFLs (NEMA, 2008). Internationally several manufacturers are producing CFLs with a mercury content of around 1 mg per lamp (Groth, 2008; Culver, 2008).

1.2. Toxicity of mercury

Mercury is a metallic element that exists in one of three forms: metallic or elemental mercury (Hg^0), inorganic mercury (Hg^+ and Hg^{2+}salts) and organic mercury (e.g. methyl mercury, phenyl mercury). Elemental mercury is a silvery liquid that can vaporize at room temperature due to its low vapor pressure (HPA, 2006) and is the form of mercury used in CFLs. The toxicology of inorganic mercury compounds and elemental mercury are briefly summarized here; organic mercury compounds are not known to be present in fluorescent lamps.[1]

When a CFL is broken, people may be exposed to elemental mercury (including vapor) and inorganic mercury compounds. The key exposure pathway to humans from broken CFLs is inhalation with 80–97% of the inhaled elemental mercury being absorbed into the body through the lungs. In comparison only 2.6% is absorbed from dermal exposure to elemental mercury vapor (HPA, 2006). Once in the body, because elemental mercury is lipid soluble, it can cross biological membranes including the blood-brain barrier and the placenta (Risher et al., 2003). Mercury is circulated throughout the body and can accumulate in the brain and the kidneys causing changes in neurological and renal function. The absorbed elemental mercury is oxidized to Hg^{2+} and is excreted in the urine (HPA, 2006). Mercury vapor has an average half-life in the body of two months, but the reported range is about 30–90 days (Risher and De Rosa, 2007).

The central nervous system is known to be the most sensitive target for exposure to mercury vapor. Exposure to mercury has caused neurological and behavioral disorders in humans (HPA, 2006), depending on the magnitude of the exposure, the exposure duration, and the age and health status of the individual, as well as the chemical species of mercury compounds (ATSDR, 1999). Humans are also known to vary in their individual susceptibility to mercury exposure (HPA, 2006) with fetuses, infants, and children under the age of six known to be sensitive subgroups (Counter et al., 2004). However, toxicity to mercury follows the expected dose–response principles; thus, at small enough doses, even sensitive members of the population are not expected to have adverse effects, as evidenced by continuing work in the Seychelles and Faroes Islands with mercury exposure to children (Schoeman et al., 2009).

Young children may also accumulate a higher internal dose of mercury vapor than adults since their ventilation rate on a body-weight basis is greater than adults. Their breathing zone is closer to the floor where mercury vapor is likely to accumulate after a CFL breakage on the floor surface (Johnson et al., 2008), because such vapor is heavier than air. Exposure scenarios discussed in this analysis take these two factors into account.

2. Data on mercury releases from broken CFLs

2.1. Fate of mercury when a CFL is broken

Once a CFL has been broken, mercury vapor, liquid mercury (if present) and mercury adsorbed onto the phosphor powder will be released (NJ MTF, 2002). It is unlikely that any spilled liquid mercury will be visible as the volume of mercury is small and any spilled mercury would form minute droplets on impact. The phosphor powder can separate from the glass when the lamp is broken (NJ MTF, 2002). The amount of mercury released as mercury vapor or associated with the phosphor powder will depend on the age of the lamp and quantity of mercury vapor in the lamp.

Fluorescent lamps will contain several species of mercury which depend on the species of the mercury added by manufacture and the age of the lamp (UNEP, 2005). Over time elemental mercury in the lamp will be oxidized and will form inorganic mercury compounds (predominantly HgO) (Aucott et al., 2003) and will partition to lamp components including the glass and phosphor powder (Jang et al., 2005). New lamps will release more mercury vapor, whereas in older or spent (used) lamps the mercury will have been oxidized and or have partitioned to lamp components. There is an initial spike in air-borne mercury concentration following breakage of a CFL or linear fluorescent tube as mercury vapor is released (Aucott et al., 2003; Environment Canada, 2004; Stahler et al., 2008), followed by slower release of mercury present in solid and liquid forms (amalgams, liquid elemental mercury, inorganic mercury and mercury absorbed onto lamp components).

Two recent studies quantified the amount of mercury released when a CFL is broken. Johnson et al. (2008) broke used and new CFLs in a 2 L Teflon container and measured the concentration of mercury vapor released over time. Two CFLs were used in the study – a 13 W lamp containing 4.5 mg of mercury and a 9 W lamp containing 5.0 mg of mercury. There was an initial high rate of mercury vapor

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[1] The following text is meant as a briefing on the toxicology of mercury and should not be considered as complete. We refer the reader to extensive analyses of this subject as further described in the studies in the reference.

release, which declined over 24-h. Over the first hour the lamps released 12–43 µg of mercury vapor (1% or less of the total amount of mercury in the lamp). During the first 24-h the 13 W lamp released 504 µg or 11.1% of the total mercury, and the 9 W lamp released 113 µg or 1.9% of the total mercury (total mercury in the lamp as specified by the manufacturer). The broken lamps continued to release mercury for at least four days (the authors did not present data beyond four days in the publication). Over four days the 13 W lamp released a total of 1.34 mg or 30% of the total amount of mercury in the lamp. Spent lamps released less mercury than new lamps. The rate of release of mercury from the broken CFLs was greater than the rate of release from a drop of liquid mercury equivalent to the amount of mercury present in the CFLs. This increased rate of release was attributed to the larger surface area of mercury adsorbed onto the phosphor and lamp components in CFLs compared to a drop of liquid mercury. The study also found that removing the glass shards following a break on carpet reduced the discharge of mercury by 67%, with the remaining mercury discharge assumed to have originated from spilled phosphor powder (Johnson et al., 2008).

Aucott et al. (2003) measured the rate of release of mercury from spent 4-ft fluorescent tubes containing 4.4 or 4.7 mg mercury. Aucott et al. (2003) noted that the mercury releases from disposal and recycling of fluorescent lamps estimated by others range widely from 1% to 80%. Aucott et al. (2003) reported that a US Environmental Protection Agency (EPA) model indicated 6%; an industry report estimated 1%; and another author estimated a range of 20–80%). The Aucott study authors assumed mercury content of 4.55 mg per tube for their experiment. The fluorescent tubes were broken inside a 32 gallon (146 L) plastic container and the concentration of mercury vapor inside the plastic container was monitored. Three room temperatures were used, 40, 60, and 85 °F for each trial. The pattern of emission was similar to that observed by Johnson et al. (2008) Initially there was a rapid rate of release followed by a declining rate of release. Aucott et al. (2003) attributed the decline in release rate to two factors – a gradual release of less volatile forms of mercury and oxidation of mercury. The mercury release rate was dependent on temperature, which was expected because of the greater volatility of mercury at higher temperatures. The authors estimated that 17–40% of the mercury present in a fluorescent tube will volatilize over a 14-day period with one-third of the mercury volatilized during the first 8-h.

Aucott et al. (2003) note that their measurements may be underestimates due to: artificial gradients in the barrel from lack of mixing of air, mercury may have adsorbed to test materials, and the Jerome 411 analyzer used measures elemental mercury vapor only. While it is believed that most of the volatile mercury in the lamps is elemental, other volatile mercury compounds and powders may be released.

Other authors (e.g., Jang et al., 2005; Raposo et al., 2004) have investigated partitioning of mercury from spent fluorescent tubes. Jang et al. (2005) measured partitioning of mercury in the vapor phase, loose phosphor, lamp glass, and end caps of fluorescent tubes. They found that total mercury concentration varied significantly among lamps, and reports that 94% or more of mercury remains either as a component of phosphor powders attached to the glass tube or in the glass, with very little mercury in the vapor phase. They concluded that the elemental mercury vapor has been partitioned to other compartments through oxidative reactions with phosphor powder and penetration mechanisms.

2.2. Maine (USA) Department of Environment Study (Stahler et al., 2008)

The most comprehensive study of mercury exposure following breakage of a CFL to date was undertaken by the Maine Department of Environmental Protection (Stahler et al., 2008). Stahler et al. (2008) focused the aim of the study to inform guidance on appropriate clean-up procedures following breakage of CFL. The study investigated a range of scenarios including clean-up method, type of lamp and floor covering (hardwood, short nap carpet and shag pile carpet). Descriptions of the 6 scenarios used in the study can be found in Table 2. Mercury concentrations were measured at 1-ft (0.3 m) and 5-ft (1.5 m) sampling heights directly above the breakage site using Lumex RA 915+ mercury analyzers, to reflect infant/toddler and adult breathing zones, respectively, in a 11′4″ × 12′1″ room with 10′ ceilings with a 30″ × 38″ window. The mercury vapor results were compared with the Maine Ambient Air Guideline (MAAG) of 300 ng/m^3 (Stahler et al., 2008), which is the same as the US EPA (2008) chronic reference concentration (RfC). The time elapsed before levels fell below the MAAG were recorded.

Table 2
Description of the six scenarios used in the Maine study (Stahler et al., 2008).

Scenario	Floor type	Clean-up	Ventilation	Hg measurements[a]
S1	Wood	None	None	Continuous until highest concentration is reached
S2	Wood	Clean-up with no vacuuming	None	Continuous
S3	Short pile rug	Clean-up with no vacuuming	None	Continuous
S4	Long pile “shag” rug	Clean-up with no vacuuming	None	Continuous
S5	Short pile rug	Clean-up with vacuuming	Yes, includes open window	Continuous, along with take discrete measurements at vacuum locations
S6	Long pile “shag” rug	Clean-up with vacuuming	Yes, includes open window	Continuous, along with take discrete measurements at vacuum locations

Note: All six scenarios used the same “Brand A” soft white A19 14 watt lamp type.
[a] All measurements were taken at 1-ft and 5-ft above site of lamp break.

For all tested scenarios, there was an initial elevated mercury concentration, which decreased very quickly and then declined more slowly over further time. Data from all the scenarios investigated indicate that it took up to about 130 min for the concentration of mercury to fall below the MAAG value of 300 ng/m^3 after removing the broken lamp and ventilating the room by opening the window. The concentrations of mercury measured above the breakage site varied depending on the sampling height (with the highest concentrations being measured at 1-ft), the brand and/or wattage of lamp, absence or presence of room ventilation, and the clean-up method. Results also varied for repeat testing of the same brand of lamp, as well as, the addition of vacuuming.

Stahler et al. (2008) identified the CFL manufacturers for the lamps they used, but did not provide the total mercury content for the CFLs. Brand A (used in scenarios S1–S6) was a 14 W lamp made by Phillips. Culver (2008) reports that Phillips CFLs sold in the United States have a mercury content of 1.23–2.7 mg/lamp.

In scenario 1 (S1) of the Maine study (Stahler et al., 2008), a single CFL (“Brand A” 60 watt equivalent) was crushed on a painted mesh cloth with mesh size 3/8″ (hardware cloth) which was on top of a piece of wood floor covering set inside a shallow open cardboard box. The window and door were closed and the ventilation system turned off. Three separate trials were run for this no clean-up scenario. Tables 3 and 4 summarize the data from scenario S1. These data are very appropriate to use for estimation of risk for scenario B in a home setting where a CFL is broken. The highest maximum

Table 3
Individual data for scenario 1 from the Maine study.

	Time before clean-up initiated (min)	Maximum concentration at 1-ft (ng/m3)	Maximum concentration at 5-ft (ng/m3)	1-h Average at 1-ft (ng/m3)	1-h Average at 5-ft (ng/m3)
Trial 1	60	8533	176	269	133
Trial 2	60	34,954	962	319	254
Trial 3	60	23,244	499	624	120

Individual trial data for Maine scenario S1 (unvented, "Brand A" 14 W = 60 W, no clean-up) were extracted from Stahler et al. (2008). One hour averages represent the average mercury concentration (ng/m3) for the first hour.

concentration measured at 1-ft was 34,954 ng/m^3 and at 5-ft was 962 ng/m^3. One hour average concentrations ranged from 120 ng/m^3 (5-ft measurement) to 624 ng/m^3 (1-ft measurement).

Points to consider with these experimental data include:

- Mercury concentrations were higher closer to the broken CFL; concentrations are not uniformly distributed in the room;
- This brand of CFL contained 1.23–2.7 mg of mercury; other lamps might have greater or smaller levels;
- These CFLs were new lamps and likely have greater mercury vapor available for immediate release versus spent lamps;
- The flooring used in scenario S1 was wood.

Table 5 summarizes the average concentrations for five Maine clean-up scenarios (S2–S6) that all utilized the same type of "Brand A" CFL. The five clean-up scenarios (S2–S6) involved different flooring materials, ventilations, and clean-up using Maine DEP existing clean-up guidance, vacuumed, or cleaned up using variations of the exiting clean-up guidance. Air was continuously sampled at 1-ft (0.3 m) and 5-ft (1.5 m) heights directly above the CFL breakage site. Results from Stahler et al. Scenarios S2–S6 are relevant to the second scenario of interest. All five of the Stahler et al. (2008) scenarios used the same brand and wattage (Brand A, 14 W) CFL, the same initial clean-up measures, and for each scenario a window was open. Three types of flooring were used (wood, short nap, and long pile carpet). Two of these scenarios (S5 and S6) also included vacuuming as part of the clean-up procedure. The results again show that measured mercury levels at the 1-ft height were consistently greater than those measured at the 5-ft height. The maximum concentration measurements and the 1-h average concentrations across the five cleaned-up scenarios were much less than for the no clean-up S1 scenario. The average maximum concentration was 11,880 ng/m^3 at 1-ft and 549 ng/m^3 at 5-ft and the 1-h average at 1-ft was 159 ng/m^3 and at 5-ft it was 90 ng/m^3. Additional tests were conducted for some of the carpeting scenarios and found that agitating the carpet (by rubbing a hand or tool over the surface) or a single vacuum session or repeated vacuuming generated peaks in the mercury concentrations. These measurements were taken at 1 in. from the carpet surface and so cannot be directly compared to the 1-ft and 5-ft measurements in the other trials.

3. Human health benchmark values

There are two agencies that have developed human health toxicity values are appropriate to use as benchmark values to compare with acute exposures to elemental mercury vapor (see Table 6). These health risk values have been developed to protect sensitive subpopulations, including the pregnant woman and her fetus, the infant, and the young child.

In the United States, Acute Exposure Guideline Levels (AEGLs) for Hazardous Substances are developed by a National Advisory Committee (NAC), which includes members from Federal Agencies, State governments, chemical industry, academia, and other organizations. AEGLs are developed for durations of 10 min to 8-h, and are defined as the threshold concentrations above which the general population, including susceptible individuals, could experience health effects of specified severities following a single airborne

Table 4
Averaged data for scenario 1 from the Maine study.

	Avg of max (ng/m^3)	RSD[a] of max (ng/m^3)	15 min avg. (ng/m^3)	15 min RSD	30 min avg. (ng/m^3)	30 min RSD	1 h avg. (ng/m^3)	1 h RSD
5-ft	546	72.4	193	50	186	50	169	44
1-ft	22,244	59.5	775	29	572	36	404	48

[a] RSD = relative standard deviation is computed as follows: RSD = 100 X (standard deviation of trial averages)/(average of trial averages). Averages mercury concentration (with relative standard deviations (RSDs)) for the three trials of Maine scenario S1 (unvented, "Brand A" 14 W, no clean-up) (extracted from Stahler et al. (2008).

Table 5
Data for scenario 2 through 6 from the Maine study.

Scenario (intake)	Avg. of max[a] (ng/m^3)	RSD of max[b]	15 min avg. (ng/m^3)	15 min RSD	30 min avg. (ng/m^3)	30 min RSD	1 h avg. (ng/m^3)	1 h RSD
S2 (5-ft)	666	23.1	106	42	67	54	37	43
S2 (1-ft)	12,261	37.7	307	43	176	41	95	42
S3 (5-ft)	770	40.9	220	19	152	21	96	19
S3 (1-ft)	8323	33.0	372	26	225	18	126	14
S4 (5-ft)	484	42.0	165	23	115	22	66	17
S4 (1-ft)	12,334	69.4	415	40	232	38	119	37
S5 (5-ft)	424	41.8	203	16	147	20	94	33
S5 (1-ft)	10,449	71.3	428	55	248	48	136	46
S6 (5-ft)	333	27.0	153	26	120	33	48	35
S6 (1-ft)	6855	127.4	251	66	154	57	87	53
All (5-ft)	549	48.1	176	33.4	132	41.7	90	57.1
All (1-ft)	11,880	77.0	42.5	52.1	266	63.3	159	82.7

Average concentrations in ng/m^3 [with relative standard deviations (RSDs)] for the three trials of Maine scenario S2–S6 (extracted from Stahler et al. (2008)).

[a] SD = relative standard deviation is computed as follows: RSD = 100 X (standard deviation of trial averages)/(average of trial averages).

[b] Three trials were run for all six scenarios. An additional trial was run for scenario 3. Two trials, S2T1 and S3T1 (1-ft intake) did not include 1-h average results due to shortened runs.

Table 6
Selected benchmark values for various times from different organizations.

Agency	Type of exposure	Value	Year	Species	Critical effect	NOAEL/LOAEL	Uncertainty factor[b]	Principle study
NAC	AEGL2[a] 10 min	3.1 mg/m^3 (3100,000 ng/m^3)	2008	Rat	Absence of fetal effects	NOAEL of 4 mg/m^3 for 2-h/day for 10 days	3 (3A, 1H)	Morgan et al. (2002)
	AEGL2 30 min	2.1 mg/m^3 (2100,000 ng/m^3)	2008	Rat	Absence of fetal effects	NOAEL of 4 mg/m^3 for 2-h/day for 10 days	3 (3A, 1H)	Morgan et al. (2002)
	AEGL2 60 min	1.7 mg/m^3 (1700,000 ng/m^3)	2008	Rat	Absence of fetal effects	NOAEL of 4 mg/m^3 for 2-h/day for 10 days	3 (3A, 1H)	Morgan et al. (2002)
	AEGL2 4-h	0.67 mg/m^3 (670,000 ng/m^3)	2008	Rat	Absence of fetal effects	NOAEL of 4 mg/m^3 for 2-h/day for 10 days	3 (3A, 1H)	Morgan et al. (2002)
	AEGL2 8-h	0.33 mg/m^3 (330,000 ng/m^3)	2008	Rat	Absence of fetal effects	NOAEL of 4 mg/m^3 for 2-h/day for 10 days	3 (3A, 1H)	Morgan et al. (2002)
OEHHA	Acute REL (1-h)	0.0018 mg/m^3 (1800 ng/m^3)	1999	Rat	CNS disturbances in offspring	LOAEL of 1.8 mg/m^3, NOAEL not observed	1000 (10L, 10A, 10H)	Danielsson et al. (1993)

[a] Note that AEGLs also incorporate a time adjustment. See appropriate NAC discussion in the text for details.

[b] Uncertainty factors are used to account for extrapolation or uncertainty in several areas. "L" accounts for extrapolation from a LOAEL to a no-observed-adverse-effect level (NOAEL); "H" accounts for inter-human variability; "A" accounts for extrapolation from experimental animals to humans; "D" is used to account for deficiencies in the available toxicity data.

exposure. An AEGL-1 is the concentration above which results in discomfort, irritation, or certain asymptomatic nonsensory effects, which are not disabling and are transient and reversible upon cessation of exposure. AEGL-2 is the concentration above which irreversible or other serious, long-lasting adverse health effects occur or may cause an impaired ability to escape. AEGL-3 is the concentration above which life-threatening health effects or death may occur.

No AEGL-1 values have been developed for mercury due to mercury having no odor or warning properties. The AEGL-2 values were developed based on studies finding no fetal effects in rats exposed to mercury vapor concentrations up to 4 mg/m^3, 2-h/day for 10 days (Livardjani et al., 1991).

The Office of Environmental Health Hazard Assessment (OEHHA) of the California Environmental Protection Agency has derived an acute Reference Exposure Level (REL) of 1800 ng/m^3 for a 1-h average exposure to mercury vapor (OEHHA, 2007). The acute REL of 1800 ng/m^3 is based on behavioral deficits measured in rats following *in utero* exposure to metallic mercury vapor (Danielsson et al., 1993). The pregnant rats were exposed to concentrations as low as 1.8 mg/m^{3}of mercury vapor for 1 h/day or 3 h/day during gestation (days 11 through 14 plus days 17 through 20). The lowest-observed-adverse-effect level (LOAEL) was 1.8 mg/m^3 for behavioral deficits in the offspring. The 1-h concentration determined by OEHHA was also 1.8 mg/m^3. The Acute REL (1-h) of 0.0018 mg/m^3 (1800 ng/m^3) was derived using an uncertainty factor of 1000 (10 for use of a LOAEL, 10 to extrapolate from rats to humans, and 10 for intra-species differences).

OEHHA released a draft revised REL for mercury in November 2007 (OEHHA, 2007). The revised Acute REL is 0.0006 mg/m^3 (600 ng/m^3) and is based on the same study and effect level, but the uncertainty factor was increased from 1000 to 3000. OEHHA increased the intra-species uncertainty factor from 10 to 30, using the default factor of 3 to account for inter-individual kinetic variability and a factor of 10 to account for greater susceptibility of children and their developing nervous systems. A factor of 30 for intraspecies uncertainty is highly unusual; typically a single uncertainty factor does not exceed 10. Due to the use of an unconventional uncertainty factor for intraspecies variability and because of the interim status of the draft Acute REL, the 0.0006 mg/m^3 value, was not considered for use as a benchmark health value for this report.

4. Results

This analysis considers two human exposure scenarios associated with the breaking of a CFL, and within each scenario, two exposure measurements that reflect different breathing areas between adults and children. The first scenario (scenario A) considers a single CFL broken in a small room that has no ventilation and no clean-up is performed. The second scenario (scenario B) also considers a single CFL broken in a small room, but in the second scenario "adequate" clean-up has been carried out and there is ventilation in the room. The second scenario also discusses the special situation of ongoing mercury release from a carpeted floor following the clean-up.

To evaluate human exposure, experimental data are preferred, especially when these data mimic the desired conditions closely, are well-controlled, and are measured reliably. However, if the conditions of the experiment do not match the desired scenario as closely as one would like, limitations and uncertainties may exist. In such cases, mathematical models are appealing as one can construct them to address parameters or measures that may not be possible in a real-world experiment. For this analysis, the experimental results from the Maine study (Stahler et al., 2008) are preferable to a simple mathematical model because the scenarios being addressed closely resemble the experimental data. These data can be compared to selected human health toxicity benchmarks.

4.1. Scenario A – Mercury emissions after no clean-up with no ventilation

The first scenario of interest is defined as a situation where a CFL is broken in an average size room and no ventilation or clean-up measures are taken. The Stahler et al. (2008) data for scenario S1 – the no clean-up scenario – are appropriate to use as an estimate of exposure for this scenario. The maximum concentrations measured in S1 are found in Table 3 and the average concentrations at 15 and 30 min and 1-h are found in Table 4. The maximum concentrations occurred within seconds of the CFL breakage.

In Fig. 1, the bars represent the average mercury vapor concentrations (from Table 4) for 15 and 30 min and 1-h and averaged maximum mercury vapor concentrations from the Maine study scenario S1 (Stahler et al., 2008) at 1-ft and 5-ft heights from the floor. These are compared with relevant short-term toxicity benchmarks (from Table 6). Please note the use of a log scale for the *y*-axis in this and other figures.

Fig. 1 illustrates that none of the three time period's average concentrations, nor the average maximum concentration at 5-ft, measured for the no clean-up scenario exceeded 15 and 30 min and 1-h AEGL-2 values. The average 1-ft concentration declines more rapidly than the 5-ft measurements. The 1-ft average maximum concentration exceeds the 1-h average exposure OEHHA

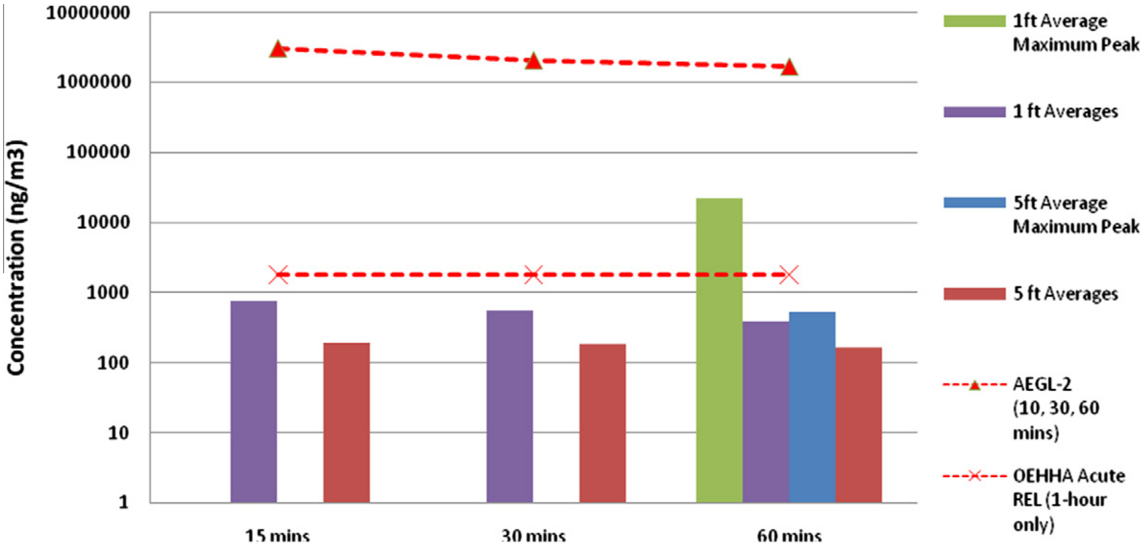

Fig. 1. Average mercury vapor concentrations at 1-ft and 5-ft from floor for three time points (15, 30 and 60 min) after breakage, and maximum peak concentrations (average of three trials from the Maine study (Stahler et al., 2008) at 1- and 5-ft represented by bars. Acute human health benchmark values are indicated with the dashed lines.

acute REL (1-h at 1800 ng/m^3), but does not exceed the AEGL-2 1-h value (1.7 mg/m^3). However, the highest 1-h average concentration at 1-ft was 624 ng/m^3 (see Table 3), well below the OEHHA acute REL of 1800 ng/m^3. In all the trials, these maximum values rapidly declined, with the 15 min average concentrations well below 1000 ng/m^3 for both heights. Other investigators (e.g., Aucott et al., 2003; OEHHA, 2007) also measured peak concentrations that rapidly declined to much lower levels in a matter of seconds or minutes.

4.2. Scenario B – Mercury emissions after clean-up with ventilation

The second scenario is a situation where a single CFL is broken in the same size room, the debris removed, and the room ventilated. This scenario is meant to represent a situation where "adequate" clean-up and ventilation measures are taken. "Adequate" clean-up includes: wearing rubber gloves or something similar; placing larger pieces into a secure closed container; sweeping up remaining pieces with a disposable broom and dustpan; using a damp rag to pat the area to remove all remaining pieces; and placing all materials into a secure container for proper disposal. A potential on-going emission from any mercury that may remain was of interest. The results from the Maine study (Stahler et al., 2008) are useful for this second situation as well. Stahler et al. (2008) measured mercury emissions from a number of scenarios where the broken CFL was cleaned-up and the room air monitored until mercury levels fell below 300 ng/m^3. Stahler et al. (2008) provides measurements of concentrations, as well as insights into concentrations levels after clean-up measures were taken. The results show that the concentrations decline very rapidly as the mercury vapor dissipates and this is supported by the experimental data from Johnson et al. (2008), Aucott et al. (2003), and others.

The second scenario of interest to New Zealand was a situation where a broken CFL was cleaned up and the room was ventilated to disperse the mercury. Stahler et al. (2008) measured mercury concentrations from a number of situations with different flooring types and clean-up procedures and measured mercury concentrations continuously until concentrations fell below 300 ng/m^3, or at least 1-h after the initial CFL breakage. Fig. 2 displays the maximum concentration (average of three trials for each scenario) and the average 1-h concentrations for scenarios S2 through S6 from the Maine study. These concentrations are compared with the 1-h average exposure OEHHA acute REL (1-h at 1800 ng/m^3) and the AEGL-2 1-h value (1.7 mg/m^3).

Fig. 3 illustrates the variation in mercury vapor released across a variety CFL brands with varying wattages. For each of these scenarios, the same clean-up method was used as scenario S2 (wood flooring) and only the type of lamp was varied (see Tables 7 and 8 for data).

4.3. Emissions remaining in carpeting

Measures to remove the glass and debris from a broken CFL eliminate much of the mercury from the room, as does ventilating the room to dissipate mercury vapor. However, with breakages of CFLs on carpeting, the mercury cannot be completely removed just by cleaning. The study by Johnson et al. (2008) found that removing the glass shards following a break on carpet reduced the discharge of mercury by 67%. They assumed the remaining mercury discharge after the broken bulb was removed originated from spilled phosphor powder (Johnson et al., 2008).

Over time, the finite amount of mercury remaining would volatilize and the available mercury will be continually depleted. Stahler et al. (2008) reported that other brand lamps produced different results.

4.4. Hazard quotients

For this screening level risk characterization, hazard quotients were developed by dividing averaged-maximum and averaged 1-h exposure concentrations by human health benchmark values. Table 9 shows hazard quotients for a no clean-up scenario (scenario A), using data from Table 4 and various benchmark values. These benchmark values are those used in the earlier figures (Figs. 1–3), and were matched as closely as possible with the exposure duration of interest.

A hazard quotient greater than 1 suggests the need to examine exposure scenario more closely as the exposure is approaching the "safe" dose. For the no clean-up scenario (scenario A) all hazard quotients were less than 1 (see Table 9), and some well below 1. This demonstrates that even without clean-up or any ventilation, the exposure to a broken CFL is not likely to pose a health risk, even if the broken lamp was not cleaned up immediately. This is confirmed if one calculates hazard quotients using the data from Stahler et al.

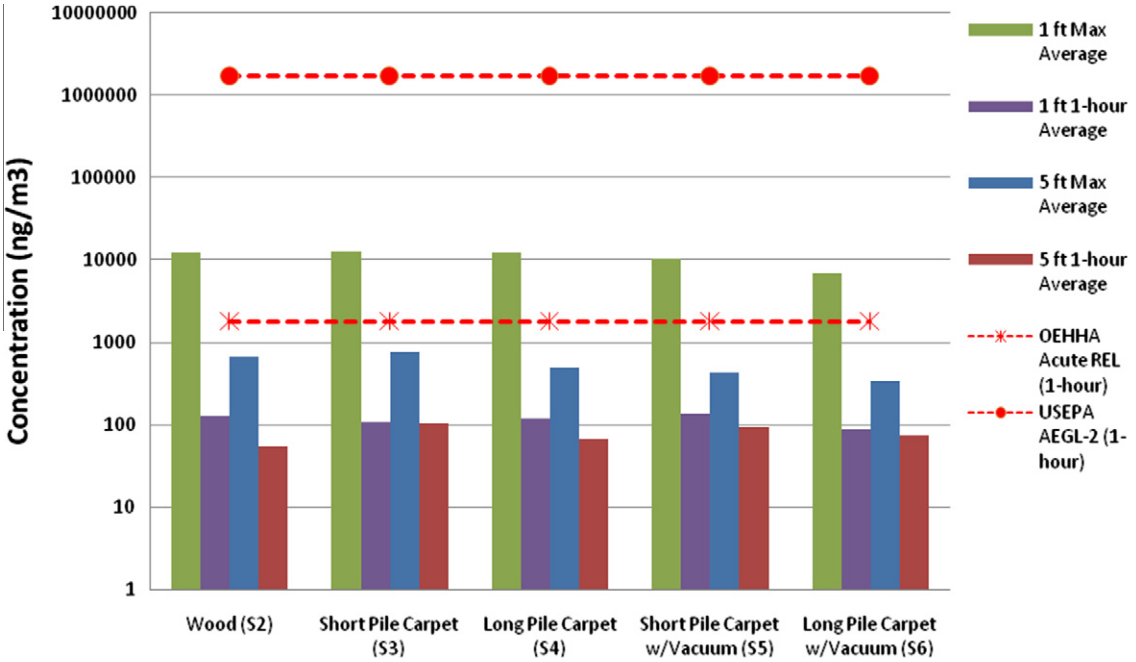

Fig. 2. Maximum and 1-h average concentrations at 1-ft and 5-ft above the floor for the Maine study (Stahler et al., 2008) scenarios 2–6 compared with the short term values of OEHHA and the US AEGL indicated with the dashed lines.

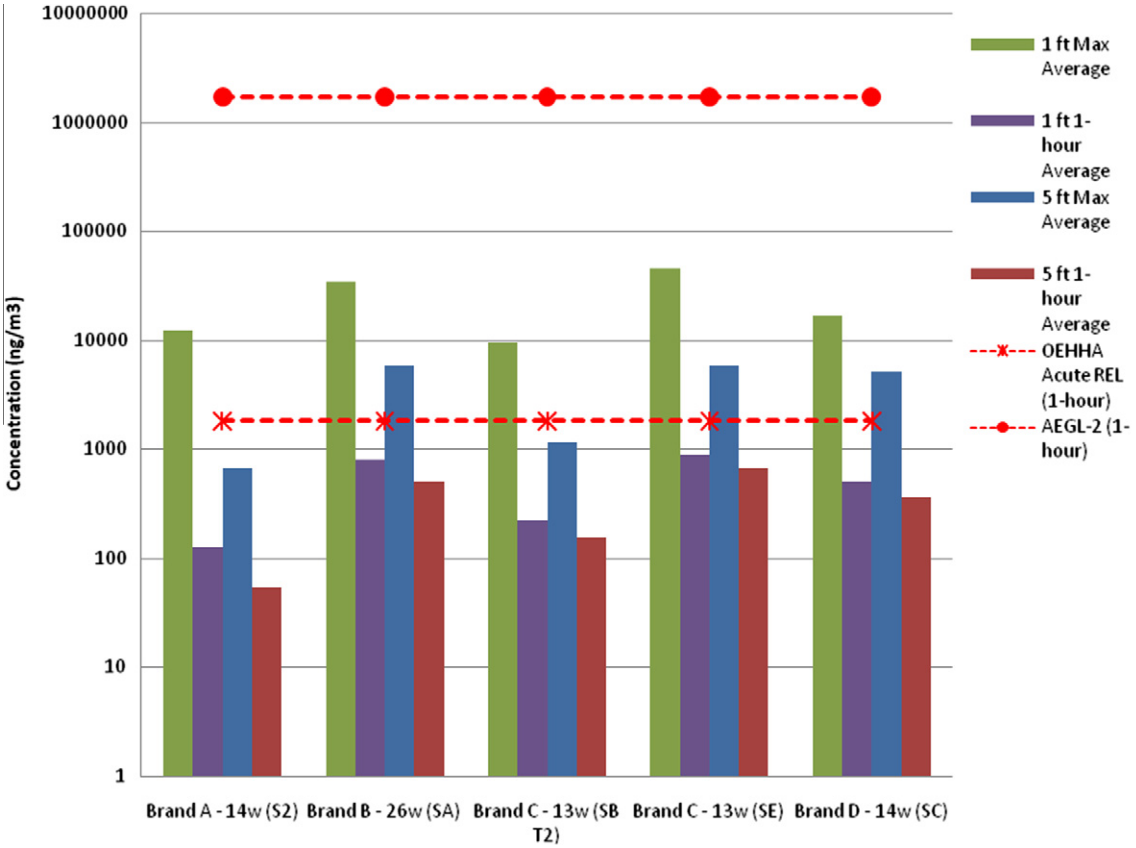

Fig. 3. Comparison of maximum and 1-h average mercury concentrations at 1-ft and 5-ft from scenarios that tested a variety of CFL Brands and Wattages from the Maine study (Stahler et al., 2008).

(2008) scenarios S2 to S6. This indicates that adequate ventilation and clean-up results in lower mercury concentrations, and like results from scenario 1, human health risk is unlikely.

Hazard quotients for various types of lamps yield similar results all under 1, even though some lamps yield up to 9-fold higher mercury releases (see Table 3).

Table 7
Comparison of individual trial data of four additional scenarios with different lamps to the results of scenario S2.

Lamp/scenario	Trial	5 ft Max	5 ft 1-h Average	1 ft Max	1 ft 1-h Average
Brand A – 14w (S2)	1	745	108	10,040	199
	2	765	26	9173	50
	3	489	29	17,569	126
Brand B – 26w (SA)	1	1640	199	7410	185
	2	9893	815	61,037	1398
Brand C – 13w (SB T2)	2	1139	155	9523	220
Brand C –13w (SE)	1	7288	527	65,094	1048
	2	4206	806	25,399	738
Brand D – 14w (SC)	1	4257	424	27,224	684
	2	5927	298	6164	310

All scenarios used wood flooring and the same clean-up methods. The wattages of the different brand lamps are identified in the text. Data extracted from Stahler et al. (2008).

Table 8
Comparison of average concentrations for each of four additional scenarios with different lamps, to the results of scenario S2.

Average of brands	5 ft Max average	5 ft 1-h Average	1 ft Max average	1 ft 1-h Average
Brand A – 14w (S2)	666	54	12,261	125
Brand B – 26w (SA)	5767	507	34,224	792
Brand C – 13w (SB T2)	1139	155	9523	220
Brand C – 13w (SE)	5747	667	45,247	893
Brand D – 14w (SC)	5092	361	16,694	497

All scenarios used wood flooring and the same clean-up methods. The wattages of the different brand lamps are identified in the text. Data extracted from Stahler et al. (2008).

Table 9
Hazard quotient for scenario A (no clean-up) based on selected benchmark concentrations.

Agency risk value	Agency value (ng/m^3)	Avg. of 1 ft max	1 ft 1-h Avg	Avg. of 5 ft max	5 ft 1-h Avg
AEGL-2 (10 min)	3100,000	<0.01	NA	<0.01	NA
AEGL-2 (30 min)	2100,000	0.01	NA	<0.01	NA
AEGL-2 (1-h)	1700,000	NA	<0.01	NA	<0.01
OEHHA Acute REL OEHHA (1-h)	1800	NA	0.2	NA	0.09

Hazard quotient = exposure estimate (or measurement)/appropriate risk value. Values greater than 1 call for additional investigation. Exposure data found in Table 4. NA = Not applicable.

5. Discussion

The human health toxicity benchmarks used for this screening assessment were selected to most closely match the exposure duration of interest. For example, even though dose response assessment values of Agency for Toxic Substances and Disease Registry (ATSDR, 1999), US Environmental Protection Agency (US EPA (2008)), and World Health Organization (WHO, 1989) are available for chronic exposure, these values were not used in this risk characterization because other dose response assessment values of more appropriate exposure duration (acute) were available.

The results (see Figs. 1–5) for these five scenarios are remarkably similar, even with the use of different types of flooring and the addition of vacuuming to the clean-up for some scenarios. Only the averaged 1-ft *maximum* concentrations exceed the OEHHA acute REL (1-h at 1800 ng/m^3) and the 300 ng/m^3 level that represents a "safe" level for a lifetime of exposure. None of the averaged maximum or 1-h concentrations exceeded the AEGL-2 (1-h value 1.7 mg/m^3) or the chronic exposure value of 300 ng/m^3. The 1-h average concentrations at either height fell below, and sometimes far below, the OEHHA 1-h REL of 1800 ng/m^3. In all of these scenarios, the maximum values rapidly declined, with all but the 15 min average concentrations below 500 ng/m^3 for either height.

The Maine study demonstrated that brands and wattage can impact the results, but the results from multiple brands are similar to Brand A used in the first six scenarios of the Stahler et al. (2008) study. While the averaged maximum concentrations generally exceeded the OEHHA acute REL, the 1-h average concentrations were all below this benchmark value. Neither the averaged maximum values, nor the 1-h average concentrations exceeded the AEGL-2 (1-h value of 1.7 mg/m^3).

Stahler et al. (2008) also utilized additional scenarios with larger wattage lamps and repeated vacuuming over a seven-day period. The results from these scenarios would be representative of ongoing mercury emissions from carpeted floors after clean-up. Figs. 4 and 5 show the mercury concentrations from a larger, 26 W CFL broken on a short pile carpet with no room ventilation on the initial day of clean-up and with vacuuming 4, 5 and 6 days after initial clean-up. For the final vacuuming (7 days after initial clean-up), the room was ventilated. Fig. 4 shows concentrations at the 1-ft height and Fig. 5 shows concentration at the 5-ft height.

These Figures demonstrate that repeated vacuuming and a higher wattage lamp, resulted in higher concentrations, although none of the averaged maximum or 1-h average concentrations exceeded the AEGL-2 value for an 8-h exposure (0.33 mg/m^3). However, several of the concentrations exceeded the OEHHA Acute REL for 1-h (of 1800 ng/m^3). These data demonstrate that repeated vacuuming over several days decreases the mercury concentrations at both measured heights, but that exposures in excess of the 1-h REL are possible, and even likely. Decreases were enhanced on day 6 and day 7, when ventilation was added to the final day of vacuuming.

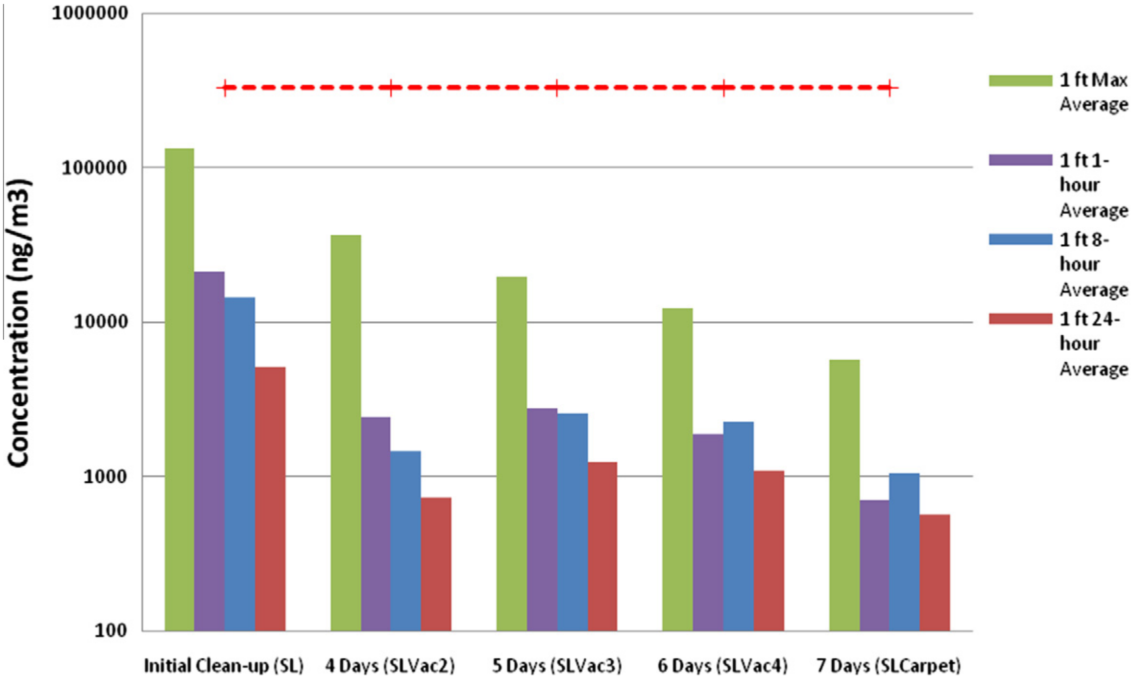

Fig. 4. Mercury concentrations at 1-ft from a scenario (SL) with multiple vacuuming sessions over a 7 day period from the Maine study (Stahler et al., 2008).

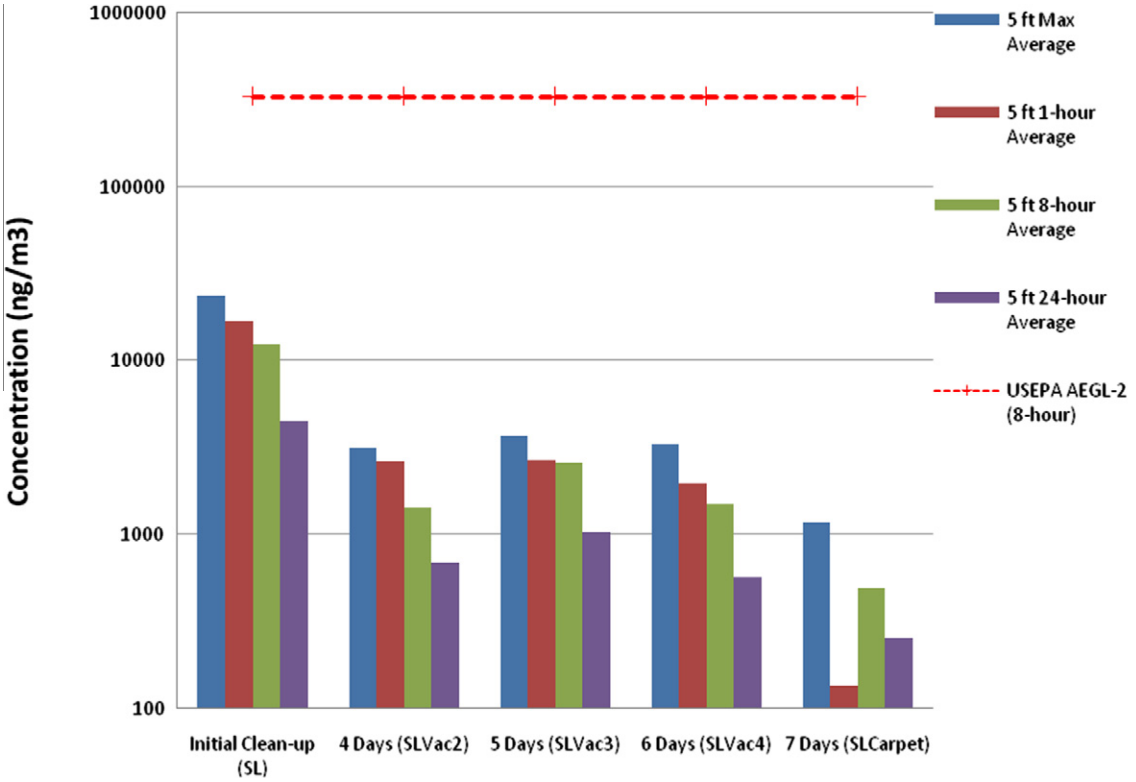

Fig. 5. Mercury concentrations at 5-ft from a scenario (SL) with multiple vacuuming sessions over a 7 day period from the Maine study (Stahler et al., 2008).

Several general areas of uncertainty are evident in this preliminary screening risk characterization. Specifically, since actual test data have been used in this risk characterization, uncertainties include:

- Mercury concentrations are higher the closer one is to the broken CFL, concentrations are not uniformly distributed in the room, and measurements at 1-ft and 5-ft heights may underestimate the occasional higher concentration elsewhere;

- The brands of CFL tested contained generally from 1 to 3 mg of mercury, other lamps might have greater or smaller levels of mercury;
- The tested CFLs were new lamps and likely have greater mercury vapor available for immediate release versus spent lamps;
- Stahler et al. (2008) illustrated that variability exists between trials within a scenario and between scenarios. This variability was not so great, however, as to affect the overall results.

Stahler et al. (2008) found that breaking one lamp on wood, short pile or shag carpet results in almost immediate high mercury concentrations, but if a window is opened and the broken lamp cleaned up, the concentrations fall below 300 ng/m^3 in 10 min or less (for the initial scenarios tested). It also appears that variations in exposure levels are primarily due to wattage. Brand A was used for the six original scenarios and it has lowest average concentrations (except brand F, which is also a low wattage lamp), but other brands with higher wattages generally resulted in greater concentrations at 15, 30 and 60 min. There are insufficient data on exposure scenarios utilizing spent lamps. However, there are data that show spent lamps release less mercury than new lamps, and therefore exposure scenarios limited to new lamps represent the worst case exposure scenario. Spent lamps are not likely to be of greater health risk when compared to the risk from new lamps.

Perhaps the largest uncertainty in this risk characterization, however, is in the choice of the toxicity benchmarks used in the development of the hazard quotients. Our choices of AEGLs of various durations for comparison with the averaged maximum 1-ft and 5-ft exposures, and of the established 1-h average exposure REL from OEHHA for comparison with the average 1-h exposures are appropriate because:

- The AEGLs and California REL have been through an external peer review process. Both sets of risk values are intended to protect sensitive individuals, including children;
- A properly developed hazard quotient should closely match lengths of exposures in both the exposure and benchmarks. It is generally not appropriate to match a 1-h exposure with a "safe" concentration for lifetime exposure, if a shorter-term value, which protects sensitive individuals, is available.

The California RELs are now being revised, with the possibility that lower values may be adopted in the near future. Moreover, a large disparity exists between the 1-h AEGL-2 of 1700,000 ng/m^3 and the 1-h REL of 1800 ng/m^3 ($\sim$900-fold). Additional investigation of this disparity should be considered a high priority. Finally, alternative benchmark values might be contemplated if the existing choices for the appropriate exposure duration of concern have irresolvable issues. For example, choices such as the EPA RfC, the ATSDR Minimal Risk Level (MRL) or the WHO Tolerable Concentration (TC), are possible, but if considered, some allowance for differences in exposure duration would be needed.

6. Conclusion

This screening assessment concludes that inorganic mercury vapor is the mercury form of concern from CFL breakage, and that releases of this mercury vapor vary within reasonably narrow bounds based on age, size and manufacturer of the lamp. Investigators have studied or measured the release of mercury vapor from CFL lamp breakage scenarios. Several of these scenarios closely match those of interest to the government of New Zealand and others, and these experimental concentrations were used to evaluate potential human health risks. A review of the data on mercury releases from these studies, along with health risk benchmark values that are conservative and matching the length-of-exposure, indicate that few situations involving breakage of CFLs will result in releases that are at levels that may generate a health concern. Some clean-up scenarios result in exposure estimates that exceed some risk characterization targets and should be further studied.

Real life conditions will vary from the experimental design used to develop these results, such as room size,[2] room temperature,[3] age of lamps, use intense vacuuming, and type of floor covering (Stahler et al., 2008). However, this variability will unlikely result in smaller hazard quotients, because the scenarios we used maximized exposures and comparisons of agency risk values were often calculated with the average maximum hourly exposures, rather than the average exposures.

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Conflict of interest statement

The authors of this manuscript have no competing interests. This manuscript was based in-part upon a project report prepared by Toxicology Excellence for Risk Assessment (TERA) under contract with the Institute of Environmental Science and Research Limited (ESR) for the New Zealand Ministry of Health (NZ MoH). The authors from TERA and NZ MoH jointly developed the manuscript. Both organizations provided funding to support the development of this manuscript.

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References

AEA Technology, 2004. WEEE and Hazardous Waste. A report prepared for DEFRA (Department for Environment, food, and Rural Affairs). AEAT/ENV/R/1688, 2004. Available online at: <http://archive.defra.gov.uk/environment/waste/producer/electrical/documents/weee-hazwaste.pdf>.

ATSDR (Agency for Toxic Substances and Disease Registry), 1999. Toxicological profile for Mercury. US Department of Health and Human Services, Public Health Service, Atlanta, Georgia.

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[2] For example, the room size used in the Stahler et al. (2008) was 39 m^3, which is larger than the 22.5 m^3 size recommended by New Zealand. However, this approximately 40% smaller room size for New Zealand will not affect the overall results, that is, this reduction only results in an increase in the maximum hazard quotient to about 0.5, which is still less than the target no-action value of 1.

[3] For example, the room temperature in the Stahler et al. (2008) study varied from 17.2 to 27.7 °C. The authors determined that correcting temperature for 23 °C (73 °F) would not significantly affect the study conclusions. However, they also concluded that if the study had been conducted at 90 °F (32 °C), the mercury concentrations would have been about 2-fold higher. This increase, while likely to happen at higher temperatures, only marginally affects the hazard quotients found in Table 7. Aucott et al. (2003) also determined that concentration increased with temperature, presumably because warmer room temperatures will result in faster evaporation of mercury.

Aucott, M., McLinden, M., Winka, M., 2003. Release of mercury from broken fluorescent lamps. J. Air Waste Manage. Assoc. 53 (2), 143–151.

Boughey, D., Webb, R., 2008. CFL Issues – Current Knowledge, Potential Solutions. Australian Government, Department of the Environment, Water, Heritage, and the Arts. Phase Out 2008 – Working Towards Global-Phase Out of Inefficient Lighting. Available online at: <http://www.energyrating.gov.au/pubs/2008-phase-out-session6-boughey.pdf>.

Counter, S.A., Buchanan, L.H., 2004. Mercury exposure in children: a review. Toxicol. Appl. Pharmacol. 198, 209–230.

Culver, A., 2008. Mercury content in lamps. Conference Presentation. EBB Conference. Mercury Containing Lamps under the Spotlight, Brussels. Available online at: <http://www.zeromercury.org/EU_developments/MercuryContent_in_Lamps.GPI.Brussels.062708.pdf>.

Danielsson, B.R.G., Fredriksson, A., Dahlgren, L., Gardlund, A.T., Olsson, L., Dencker, L., Archer, T., 1993. Behavioral effects of prenatal metallic mercury inhalation exposure in rats. Neurotoxicol. Teratol. 15, 391–396.

Energy Star, 2008. Frequently Asked Questions: Information on Compact Fluorescent Light Lamps (CFLs) and Mercury. US Environmental Protection Agency and US Department of Energy. Available online at: <http://www.energystar.gov/ia/partners/promotions/change_light/downloads/Fact_Sheet_Mercury.pdf>.

Environment Canada, 2004. Mercury and the Environment – Sources of mercury – Mercury containing products. The Green Lane – Environment Canada's World Wide Web site. Available online at: <http://www.ec.gc.ca/MERCURY/SM/EN/sm-mcp.cfm>.

Groth, E., 2008. Shedding light on mercury risks from CFL breakage. The Mercury Policy Project. Available online at: <http://mpp.cclearn.org/wp-content/uploads/2008/08/final_shedding_light_all.pdf>.

HPA (Health Protection Agency), 2006. Inorganic mercury/elemental mercury. Toxicological Overview. HPA, United Kingdom. Available online at: <http://www.hpa.org.uk/web/HPAwebFile/HPAweb_C/1194947331729>.

Jang, M., Hong, S.M., Park, J.K., 2005. Characterization and recovery of mercury from spent fluorescent lamps. Waste Manage. 25, 5–14.

Johnson, N.C., Manchester, S., Sarin, L., Gao, Y., Kulaots, I., Hurt, R.H., 2008. Mercury release from broken compact fluorescent lamps and in situ capture by new nanomaterial sorbents. Environ. Sci. Technol. 42, 5772–5778.

Livardjani, F., Ledig, M., Kopp, P., Dahlet, M., Leroy, M., Jaeger, A., 1991. Lung and blood superoxide dimutase activity in mercury vapor exposed rats: effects of n-acetylcysteine treatment. Toxicology 66 (3), 289–295.

Morgan, D.L., Price, H.C., Fernando, R., Chanda, S.M., O'Conner, R.W., Barone, S.S., Herr, D.W., Belilies, R.P., 2002. Disposition of inhaled mercury vapor in pregnant rats: maternal toxicity and effects on developmental outcome. Toxicol. Sci. 66 (2), 261–273.

NEMA (National Electrical Manufacturers Association), 2008. NEMA Standards Publication LL 8-2008: Limits on Mercury Content in Self-Ballasted Compact Fluorescent Lamps. National Electrical Manufacturers Association. Available online at: <http://www.nema.org/media/pr/20070313a.cfm>.

NEWMOA (Northeast Waste Management Officials Association), 2008. Mercury Use in Lighting. Factsheet. Northeast Waste Management Officials' Association, Boston. Available online at: <http://www.newmoa.org/prevention/mercury/imerc/FactSheets/lighting.cfm>.

NJ MTF (New Jersey Mercury Task Force), 2002. New Jersey Mercury Task Force Volume III: Sources of Mercury to New Jersey's Environment. Department of Environmental Protection, New Jersey. Available online at: <http://www.nj.gov/dep/dsr/mercury_task_force.htm>.

OEHHA (Office of Environmental Health Hazard Assessment), 2007. Mercury Reference Exposure Levels. Public Review Draft November 2, 2007. Office of Environmental Health Hazard Assessment, State of California. Available online at: <http://oehha.ca.gov/air/hot_spots/pdf/MercuryPR.pdf>.

Parsons, D., 2006. The environmental impact of compact fluorescent lamps and incandescent lamps for Australian conditions. Environ. Eng. 7 (2), 8–14.

Raposo, C., Windmöller, C.C., Junior, W.A.D., 2004. Mercury speciation in fluorescent lamps by thermal release analysis. Waste Manage. 23, 879–886.

Risher, J.F., De Rosa, C.T., 2007. Inorganic: the other mercury. J. Environ. Health 70 (4), 9–16.

Risher, J.F., Nickle, R.A., Amler, S.N., 2003. Elemental mercury poisoning in occupational and residential settings. Int. J. Hyg. Environ. Health 206, 371–379.

RoHs (Restriction of the Use of Certain Hazardous Substances in Electrical and Electronic Equipment Regulations), 2008. RoHS Regulations. Department for Business, Enterprise & Regulatory Reform (BERR), Government Guidance Notes. Available online at: <http://www.bis.gov.uk/nmo/enforcement/rohs-home>.

Schoeman, K., Bend, J.R., Hill, J., Nash, K.M.A., Koren, G., 2009. Defining a lowest observable adverse effect hair concentrations of mercury for neurodevelopmental effects of prenatal methylmercury exposure through maternal fish consumption: a systematic review. Ther. Drug Monit. 31 (6), 670–682.

Singhvi, R., Patel, J., Kansal, V., Gasser, C., Kalnicky, D., 2008. Total Metallic Mercury Analysis in Compact Fluorescent Lamps – Sample Preparation and Analysis. Poster Presentation. 24th Annual National Environmental Monitoring Conference, 11–15 Aug, Washington, DC.

Stahler, D., Ladner, S., Jackson, H., 2008. Maine Compact Fluorescent Lamp Study. Department of Environmental Protection, Maine. Available online at: <http://maine.gov/dep/rwm/homeowner/cflreport.htm>.

UNEP (United Nations Environment Programme), 2005. Toolkit for identification and quantification of mercury releases: Pilot Draft Inter-Organisation. Programme for the Sound Management of Chemicals, United Nations Environmental Programme. Available online at: <http://www.chem.unep.ch/mercury/Toolkit/default.htm>.

US EPA (Environmental Protection Agency), 2008. Mercury, elemental (CASRN 7439-97-6). Integrated Risk Information System (IRIS), United States Environmental Protection Agency, Washington, DC. Available online at: <http://www.epa.gov/ncea/iris/subst/0370.htm>.

World Health Organization (WHO). 1989. Mercury – environmental aspects. Vol. 86. Geneva, Switzerland: World Health Organization, International Programme on Chemical Safety. Available online at: <www.who.int/ipcs/publications/cicad/en/cicad50.pdf>.