Scientific Committee on Health and Environmental Risks

SCHER

Opinion on Mercury in certain
Energy-saving Light Bulbs – Exposure of Children

The SCHER adopted this opinion at its 16th plenary on 22 March 2012

About the Scientific Committees

Three independent non-food Scientific Committees provide the Commission with the scientific advice it needs when preparing policy and proposals relating to consumer safety, public health and the environment. The Committees also draw the Commission's attention to the new or emerging problems which may pose an actual or potential threat.

They are: the Scientific Committee on Consumer Safety (SCCS), the Scientific Committee on Health and Environmental Risks (SCHER) and the Scientific Committee on Emerging and Newly Identified Health Risks (SCENIHR) and are made up of external experts.

In addition, the Commission relies upon the work of the European Food Safety Authority (EFSA), the European Medicines Evaluation Agency (EMEA), the European Centre for Disease prevention and Control (ECDC) and the European Chemicals Agency (ECHA).

SCHER

Opinions on risks related to pollutants in the environmental media and other biological and physical factors or changing physical conditions which may have a negative impact on health and the environment, for example in relation to air quality, waters, waste and soils, as well as on life cycle environmental assessment. It shall also address health and safety issues related to the toxicity and eco-toxicity of biocides.

It may also address questions relating to examination of the toxicity and eco-toxicity of chemical, biochemical and biological compounds whose use may have harmful consequences for human health and the environment. In addition, the Committee will address questions relating to methodological aspect of the assessment of health and environmental risks of chemicals, including mixtures of chemicals, as necessary for providing sound and consistent advice in its own areas of competence as well as in order to contribute to the relevant issues in close cooperation with other European agencies.

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1. BACKGROUND

Certain energy-saving light bulbs, namely compact fluorescent lamps (CFLs), are widely available on the market and are offered for saving electricity and, eventually, reducing carbon dioxide emissions particularly from coal-fired power plants. They fulfil the requirements of Commission Regulation (EC) No 244/2009 on ecodesign requirements for non-directional household lamps[1] (Ecodesign Regulation), in contrast to traditional incandescent light bulbs which will be phased out progressively in accordance with the Regulation.

The Scientific Committee on Health and Environmental Risks (SCHER) provided an opinion on mercury released from breaking CFLs (in May 2010)[1], but could not conclude on the potential risk of children due to lack of data. SCHER is now asked for an opinion on the potential mercury exposure to children, and thus the risk.

2. TERMS OF REFERENCE

Against the above background, taking into account all available scientific assessments on mercury, including the Risk Assessment under 793/93/EEC and the previous opinions of SCHER, CSTEE, SCENIHR and the EFSA Scientific Panel on Contaminants in the Food Chain, the SCHER is requested to assess the possible health risks to children, from the mercury released from accidental breakage of CFLs.

3. OPINION

Toxicology of inorganic mercury

The environmental benefits of introducing CFLs for lighting due to a reduced Hg-emission from fossil fuel use for energy production has been assessed in a previous opinion of SCHER (2010). Aspects of the hazard assessment for inorganic and elemental mercury have been summarized in the previous opinion and are described in detail in a number of monographs (ATSDR, 1997; Clarkson and Magos, 2006; IRIS, 2002; UBA, 2011; US-EPA, 2010; WHO/IPCS, 2002). A comprehensive review on mercury exposure in children is available (Counter and Buchanan 2004). Other relevant papers have also been published addressing the issue of critical windows of vulnerability during fetus and child development (ATSDR 1999, CDC 1991, WHO 2003).

IPCS has set a tolerable (oral) daily intake (TDI) for lifetime exposure to elemental and inorganic mercury of 2 µg/kg bw/day. The TDI also covers sensitive subgroups such as children (WHO/IPCS, 2002). Other derivations of tolerable oral intakes often refer to the much more toxic methyl mercury ion, which is not relevant in this context.

When comparing guideline values for mercury, differences in bioavailability after inhalation and oral exposures and differences in distribution of elemental Hg and Hg-ions need to be considered. After inhalation, elemental Hg is more bioavailable as compared to oral intake, but absorbed elemental Hg is oxidized in blood to Hg-ions, which cannot readily penetrate biological membranes. Mercury may cross the placenta and blood-brain barriers and may thus be deposited in brain and can be delivered to the fetus (Morgan et al 2002, Fredriksson et al. 1992 and 1996, Danielsson et al. 1993, WHO 2010, Clarkson et al. 2007, Counter and Buchanan 2004).

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[1] http://ec.europa.eu/health/scientific_committees/environmental_risks/docs/scher_o_124.pdf

While many guideline values are available for tolerable exposures to Hg by repeated inhalation exposures (ATSDR, 1997; Clarkson and Magos, 2006; IRIS, 2002; UBA, 2011; US-EPA, 2010; WHO/IPCS, 2002), only the US EPA has derived regulatory values ("acute exposure guideline values", AEGLs) for single exposures of up to eight hours to assess a single inhalation exposure to elemental Hg as expected after breaking a CFL (see table 1).

Table 1: Acute exposure guideline values for Hg in air for different durations of inhalation exposures in humans derived for single exposures and different time scales

Organisation	Hg Concentration (mg/m^3)	Extrapolation factor	Comments
US EPA AEGL-2 (US-EPA, 2010)	3.1 (10 min) 1.7 (60 min) 0.67 (4 h) 0.33 (8 h)	Interspecies extrapolation factor of 10 and adjustment for duration of exposure	Based on NOAEL (4 mg/m^3) from developmental toxicity study in rats after inhalation of elemental Hg

The US EPA AEGL-2 is defined as "*the airborne concentration (expressed as mg/m³) of a substance* (for a single exposure) *above which it is predicted that the general population, including susceptible individuals, could experience irreversible or other serious, long-lasting adverse health effects*.

Regarding breakage of a CFL, the US EPA AEGL-2 values are the most appropriate limit value to evaluate single inhalation exposures to elemental Hg. The AEGL-values are based on a reasonable extrapolation from a reproductive toxicology study (Morgan et al., 2002).

Indeed, to derive AEGLs, the US EPA used the NOAEL from a developmental toxicity study in rats with maternal exposures for 2 hours/day for 10 days and a NOAEL of 4 mg/m^3. Developmental effects including increased resorptions, decreased litter size and decreased neonatal weight were observed at the next higher concentration of 8 mg/m^3 (Morgan et al., 2002). A factor of 10 was used for the interspecies uncertainty; this represents a conservative approach since rodents have a higher respiratory rate and cardiac output compared with humans resulting in faster uptake. The conservatism of the obtained values also includes the facts that the exposures in the study from which the NOAEL was derived were repeated for 10 days and the AEGL-2 is only assessed for exposures up to 8 h and that human monitoring studies show effects at Hg-concentrations of 0.4 to 2 mg/m^3 (above the AEGL-value for a single 8-hour exposure) only after continued exposures for many years.

Experimental data on Hg concentrations in indoor air after breaking a CFL

Several recent studies have measured the time-course of Hg-concentrations in the air of representative rooms under controlled conditions after breaking different types of CFLs and have assessed the impact of venting and cleanup on these concentrations.

In the US, the state of Maine performed a comprehensive series of experiments breaking CFLs containing up to 5 mg of Hg in a room with a volume of approx. 32 m³

at temperatures between 19 and 23°C (ME-EPA, 2008). Different floor types (rug and prefinished hardwood) and cleanup procedures were used. Hg concentrations were continuously monitored. Information of air exchange rates in the room was not provided. Without venting or cleanup, peak Hg air concentrations at 30 cm from the broken CFL (breathing zone of a small child) were determined as 35 μg/m^3 with one-hour averages up to 0.62 μg/m^3. The peak concentrations were only present at the initial sampling points within 8 minutes after breakage. The air concentrations of Hg rapidly declined after the 8-minute sampling point. Venting the room and performing specific cleanup operations reduced peak concentrations of Hg at 30 cm (see above) to < 13 μg/m^3 and one-hour averages to < 0.3 μg/m^3 in half of the trials. In one experiment, peak air concentrations of up to 61 μg Hg/m^3 and a one-hour average of 2.7 μg/m^3 were obtained with a different lamp type (26W) with cleanup. Increases in room temperature resulted in higher release rates of Hg from broken CFLs; at a room temperature of 32 C°, air concentrations of Hg were twice as high as at 23° C.

The German Umweltbundesamt (UBA) commissioned two studies to determine Hg air concentrations after breaking of a CFL (Fraunhofer-WKI, 2011a, b). In the first study (Fraunhofer-WKI, 2011b), four different CFLs (one with protective bulb over the lighting coil) containing 1.5 – 2 mg Hg were broken in a room of 24 m^3 at 23°C with an air exchange rate of 0.5/h. Both hot and cold CFLs were included and breakage was performed both over carpet and hardwood floors. The results on Hg air concentrations and their time courses were highly inconsistent. In some experiments, time-courses with short peaks of Hg and a rapid decline similar to those reported in the Maine study were observed. However, several of the experiments resulted in Hg air concentrations of 8 μg/m^3 for several hours with only a slow decline. The authors assume that differences in lamp design and exact site of damage on the lamp may have a major impact on the extent of Hg release from a broken CFL.

In the second study (Fraunhofer-WKI, 2011a), UBA assessed the effect of two different scenarios of cleanup and venting the room after breaking only one type of CFL. The cleanup procedure consisted in opening the windows, collection of lamp debris and vacuum cleaning. The CFLs were broken in an office room of 68 m^3 over carpet or hardwood floor and Hg was also measured in air at a distance of 30 cm from the debris. Immediately after breaking the CFL, the window was opened. This study showed short spikes in Hg air concentration of up to 6 μg/m^3 for a maximum of four minutes. Hg then returned to < 0.035 μg/m^3 at the next four minute sampling point and one hour average air concentrations were approximately 0.4 μg/m^3 (Fraunhofer-WKI, 2011a). Some of the results have been included in a publication, which also reports results on the air concentrations of Hg in an office room when breaking both hot and cold CFLs over different flooring materials with rapid venting of the room after the breakage of the CFL. They report short peak concentrations of up to 5.7 μg/m^3 which declined within 10 – 15 minutes to < 0.04 μg/m^3 (Salthammer et al., 2012).

A CFL producer also reported peak concentrations between 30 (cold) and up to 700 μg/m^3 (hot) at 30 cm from the debris in a room of 39 m^3 (air exchange rates not given) after breaking CFLs, venting was also very efficient to reduce peak levels and duration (Gorbacheva, 2011).

The Landesamt für Gesundheit und Lebensmittelsicherheit of the State of Bavaria (LGL) in Germany assessed Hg concentrations in air after breaking CFLs with Hg contents between 1.4 and 4 mg at temperatures between 14 and 21°C. Hg in air was continuously monitored at a distance of 1 m from the broken CFL in a room with a very low air exchange ratio of 0.1/hour. Room air was mixed by a ventilator. The peak concentrations of Hg in air were determined to be between 0.4 and 0.9 μg/m^3 at room temperatures between 14 and 18° C. At 22°C, the peak air concentrations

were between 0.68 and 10.3 µg/m^3. Breaking a hot CFL gave significantly higher average room concentrations (5 µg/m^3). In the scenario with very low air exchange, concentrations of Hg remained high over 72 h (reduced to only approx. 50% of peak concentration right after breakage). However, peak concentrations of Hg were significantly lower at an air exchange ratio of 2/hour (Fromme et al., 2011).

None of these publications assessed the extent of condensation of Hg to room walls or furniture and the possible transfer into dust, but it is unlikely that all (liquid) Hg liberated by a broken lamp and deposited on a surface will transform into Hg vapour within a couple of days, as has been shown in a theoretical and experimental study (Winter 2003). The SCHER also evaluated several other references that describe the evaporation of Hg vapour from liquid Hg drops on surfaces, but considered these as providing no additional information for the subject (Johnson et al., 2008; Winter, 2003).

Exposure assessment based on measured data

The measured data show that Hg concentrations in indoor air may be highly variable depending on type of CFL broken, temperature of the lamp and room temperature, site of breakage of the lamp body, time of use of the CFL, ventilation conditions and type of flooring.

Therefore, for exposure assessment, the SCHER evaluated three scenarios based on the highest measured values of Hg in the breaking experiments and applied conservative extrapolations. The measured air concentrations are then compared to the AEGL-2, the most appropriate limit value to evaluate inhalation exposures to elemental Hg deriving from a CFL breakage.

The predicted exposures were also translated to intakes based on child-specific inhalation rates and compared with the oral TDI (corrected for differences in systemic bioavailability between oral and inhalation exposures) which is the most commonly used reference value to assess and compare exposures to Hg from various sources for lifetime exposures. However, the TDI is not particularly relevant for the specific exposure scenarios following a CFL breakage, since it refers to a life-long repeated daily exposure which cannot be directly applied to the 0.5 - 5 day peak exposure scenario following the event of a broken CFL.

Scenario 1: This scenario uses the Hg concentrations measured in a room with a very low air exchange, without cleanup and a very long exposure time of 36 h. This scenario has been determined by the UBA and the LGL as an extreme case (Fromme et al., 2011; UBA, 2011).

Scenario 2: This scenario is based on the time-course of Hg in air determined in one of the many experiments in the Maine study (ME-EPA, 2008). This scenario, despite venting and cleanup, gave the highest average one-hour Hg-concentration. In the Maine study, the one-hour averages declined significantly over several hours. Thus, the assumption that the one-hour average is representative of a four-hour exposure adds additional conservatism.

Scenario 3: This scenario uses the peak Hg-concentration in air when a CFL is broken in a room (Salthammer et al., 2012) with immediate venting and the exposure assessment implies that children and pregnant females leave the room within one minute to derive realistic potential exposures.

In the measurements forming the basis for scenario 1, peak concentrations of 10 µg Hg/m^3 are reached and high Hg concentrations are present for several days due to an unusually low air exchange rate (Fromme et al., 2011). The long-term measurements performed by LGL indicate a linear reduction of Hg air concentrations to 50% of the Hg-peak within approximately 36 h. In this case, an average air concentration of app. 8 µg Hg/m^3 is present over 24 hours. This value remains below

the US-EPA AEGL-2 value even when an additional time scale adjustment is performed to account for a inhalation exposure duration up to 36 h (giving a tolerable concentration of 50 µg Hg/m^3 for this time period). Converting the average air concentration of 8 µg Hg/m^3 with an exposure duration of 36 h to a systemic Hg dose using a child specific inhalation rate of 0.5 m^3/h and a lung absorption of 80% gives a dose of 10 µg/kg bw/day for a 7 kg child (US-EPA, 2011) delivered over these two days.

In scenario 2, the highest average one-hour Hg-concentration measured was 3 µg/m^3. Exposure to this concentration is assumed to be present for four hours which adds conservatism to scenario 2. This concentration is well below the AEGL-2. Using the child-specific parameters from above, the systemic dose is calculated to be 0.6 µg Hg/kg bw for one day.

In scenario 3, peak Hg concentrations of 10 µg/m^3 are present for app. 20 min, the high concentration then dropped to < 0.05 µg/m^3 and remained at this level when venting was repeatedly performed. Since children and pregnant women leave the room in this scenario within one minute. A one minute exposure to this peak will result in a single dose of 0.006 µg Hg/kg bw for a 7 kg child. When integrating the highest air concentration determined in the breakage studies (700 µg/m^3), intake in child will be approximately 0.6 µg Hg/kg bw.

The further fate of the airborne Hg released into a room by breaking a CFL has not been assessed experimentally. Therefore, an assessment of indirect exposures of children due to ingestion of deposited Hg by hand-to-mouth contact cannot be supported by measured concentrations of Hg in dust or on room surfaces. The Hg released from a lamp in gaseous form under conditions of low air exchanges will remain in room air for a prolonged time and may in part adsorb to the surfaces of walls, furniture, and other items present in the room. Extent of adsorption of Hg to different surface materials varies widely and is depending on many parameters (Spedding and Hamilton, 1982). Transfer of absorbed Hg from walls and furniture to dust is considered low, based on measured content of Hg in airborne dust in buildings with considerable Hg spills (Gioda et al., 2007). While the dermal absorption of elemental Hg is very low, the amount possibly ingested from dermal contamination of a child by hand to mouth contact is also not defined. Hg remaining in the debris of the lamp may be moving to other parts of the room due to the tendency of Hg drops to concentrate in sinks. Therefore, an assessment of indirect exposure due to dust consumption or hand-to-mouth contact in children is highly uncertain. If breakage is followed by immediate venting (which will release most of the airborne Hg to the outside air), removal of the lamp debris, and having children and pregnant women leave the room, such an assessment will not be relevant for potential children exposures since Hg released to air will be rapidly removed from the room by ventilation. Hg remaining in the debris of the CFL will also be removed.

Conclusions

Comparing the Hg concentrations determined in representative rooms, all values are well below the recently derived AEGL-2 values.

When comparing intakes by inhalation with the TDI, differences in systemic availability of Hg after inhalation and oral intake need to be considered and the TDI needs to be adjusted accordingly. Absorption of inorganic Hg after oral intake is only app. 8 % of the Hg absorption after inhalation and thus requires a correction factor of 10 to the oral TDI, a tolerable lifetime intake for elemental Hg adjusted for inhalation exposures would thus be 0.2 µg/kg bw/day (ATSDR, 1997; US-EPA, 2010).

When comparing the intakes in the different scenarios with the tolerable daily intake (TDI) adjusted to inhalation exposures, Hg concentrations in indoor air will result in intakes of Hg which will exceed the TDI. In the extreme scenario 1 with very low air

exchange ratios, no venting, no cleanup, and exposure duration for 36 h, the Hg intake will exceed the TDI by 50 fold. In scenario 2 and in scenario 3 (only when using the extremely high value of 700 µg/m^3 with ventilation, clean up of the room) the Hg intake will be 3 times above the TDI. Rapid venting of the room as in scenario 3 immediately after the breakage of a CFL, and having pregnant females and small children leaving the room, resulted in Hg intakes well below the TDI. Hence, venting the room immediately after the breakage of a CFL, asking pregnant females and small children to leave the room, and collecting of the lamp debris in appropriate containers significantly reduces the peak and one-hour average Hg-concentrations and the exposure of potentially sensitive population groups to peak concentrations of Hg. Due to the rapid removal of Hg from the room when high air exchanges are used, adequate venting and debris removal will also minimize the potential for indirect exposures or rebound effects due to evaporation of deposited Hg.

On the other hand, socioeconomic factors (e.g. living under poor, cramped conditions), and climatic conditions (e.g. during winter) may influence the ability of consumers to leave the room and/or carry out a proper ventilation when a CFL breaks. In addition, some studies have shown that using a vacuum cleaner for the removal of debris when a CFL breaks, may prolong the exposures to the released Hg due to the creation of aerosols by the aspiration functioning of the vacuum cleaner.

The SCHER considers that short peak inhalation exposures to peak Hg-concentrations in air occurring as a result of accidental breakage of CFLs and intakes of Hg above the TDI for a very limited time are unlikely to pose a health risk for the following reasons:

- the relevant health effects of Hg (neurotoxicity) are not depending on dose rates when delivered by short peaks, but on cumulative dose received due to the long half-life of Hg in the mammalian body (ATSDR, 1997; US-EPA, 2010).

- JECFA derives provisional weekly intakes for persistent contaminants such as Hg. EFSA derives weekly or monthly tolerable intakes as the daily intake cannot be considered representative. The monthly/weekly intakes are time-weighted averages and may include spikes in exposures (e.g. methyl mercury from a two fish meals per week may exhaust the tolerable weekly intake) (EFSA, 2004).

- The placental passage and thus the exposure of the fetus to Hg is lower than that of the maternal organisms. In a developmental toxicity study, mean concentrations of Hg in the brain of dams exposed by inhalation to 4 mg Hg/m^3 for 10 days were up to 1,000-higher in the kidney and 60-fold higher in brain as compared to the fetus (Morgan et al. 2002).

- High air concentrations of Hg are not directly translated to spikes in blood, fetal, and brain concentrations of Hg due to kinetics of intake and distribution. Development of peak Hg-concentrations is comparatively slow. In humans, after a 15 min inhalation exposure to Hg, peak blood levels were only reached 5 hours after the inhalation exposure (Sandborgh-Englund et al., 1998). Distribution of Hg to organs is also slow since peak levels of Hg in tissues after inhalation exposures are reached within 24 hours, except in the brain where peak levels are achieved within 2–3 days (Hursh et al., 1976).

- Consistently, the kidney is the major organ of Hg disposition after inhalation. After inhalation exposure of dams, most of the fetal Hg was

present in liver and kidney. Concentrations of Hg in liver and kidney were also higher then in brain in neonatal rats exposed to Hg by inhalation (ATSDR, 1997; US-EPA, 2010).

-   Since a single oral intake of metallic Hg at much larger amounts as compared to that released from a CFL (fractured Hg based thermometer) is not considered a risk (Sims-Williams, 1976), it is unlikely that indirect intake of Hg released into the room from a broken CFL and deposited on room surfaces will represent a health risk, specifically when venting is performed quickly

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