

# A Ban on Incandescent Lamps? An Analysis of the Proposed NRCan Lighting Standard

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*Natural Resources Canada (NRCan) has proposed to “phase-out inefficient lighting in common uses by 2012”, where “common uses” refers to general service incandescent lamps (40 to 100 watts). To do this it has proposed a Tier 1 efficacy (lumens per watt) standard roughly 50% higher than the best fit curve of current efficacies for incandescent lamps as a function of total lumen output. Proposed Tier 2 standards, scheduled to go into effect in 2015, represent a 100% increase from current efficacies. At this time the most efficient commercially available technology for general purpose residential lighting in the 40 to 100 watt range is compact fluorescent lamps (CFLs). A model was developed using the Canadian housing stock, lighting profile and provincial electricity generation characteristics to evaluate the energy use and greenhouse gas emissions of a widespread switch from incandescent lamps to CFLs. The disposal and power plant emission effects of mercury were evaluated in terms of toxicity potential. Our findings indicate that replacing 90% of 40 to 100 watt general service incandescent lamps with CFLs could reduce residential lighting electricity use by 12.5 TWh, a 58% reduction from 2003 use. After considering the effect on space conditioning systems, the net reduction in the residential sector will be 37% (8.0 TWh/yr) of the baseline scenario. Greenhouse gas (GHG) emissions from the residential lighting sector would be reduced by 46% (2.7 MtCO<sub>2e</sub>/yr). The effects of mercury include: a decrease of 28% on human toxicity potential, a 34% decrease in marine sediment and marine aquatic ecotoxicity potentials, an increase of 259% in freshwater aquatic ecotoxicity potential increases, an increase of 264% in freshwater sediment ecotoxicity potential increases and an increase of 195% in terrestrial ecotoxicity potential. The regulation would reduce the annual cost of electricity for lighting per household by CDN\$85. Taking into consideration annual lamp replacement costs and additional heating fuel costs resulting from the regulation, the average Canadian home could see a reduction of CDN\$73 per year in costs associated with lighting.*

## 1. INTRODUCTION

Electric lighting is a ubiquitous feature of the built environment. The ability to provide light absent the sun is needed for economic productivity and human safety as well as convenience. However, residential and commercial lighting in Canada account for roughly 5% and 10% of energy use in those sectors, respectively (NRCan, 2006). With growing concern regarding the link between energy production and global climate change, more attention is being directed at reducing energy end use. To this end, Natural Resources Canada (NRCan) has proposed a regulation that would require higher energy efficiency in general service lighting (NRCan, 2007b).

The authors evaluated the effects of adopting compact fluorescent lamps as the dominant general service lamp in the Canadian residential sector. The analysis was based on the requirements of NRCan’s proposed regulation to phase-out lamps below increasingly stringent efficiency thresholds. A Tier 1 efficacy (measured in lumens/Watt) standard, scheduled to take effect in 2012, represents a roughly 50% increase in the best fit curve of current efficacies for incandescent lamps as a function of total lumen output. Tier 2 standards are scheduled take effect in 2015 and represent a 100% increase over current efficacies. This assessment mainly pertains to energy and environmental issues, although consumer cost-benefit is considered.

Compact fluorescent lamps (CFLs) were chosen as the primary replacement technology because they already meet the proposed requirements (Waide, 2007) and are a “drop-in” replacement for general service incandescent lamps. Replacing incandescent lamps with CFLs would reduce energy use. The effect on GHG would vary greatly by province due to difference in electricity generation. For example Alberta generates 97% of its electricity from GHG-intensive sources such as coal and natural gas. Quebec’s generation mix is 96% hydro with low GHG emissions (CEA, 2006). Upstream impacts (e.g., manufacturing and shipping) are

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less influential relative to the use phase (Parsons, 2006: 11; Pfeifer, 1996: 10) and therefore will be excluded from this study.

Less than 10% of the electricity supplied to an incandescent lamp is converted to light, the balance being heat. This reduces heating requirements in winter and increases cooling requirements in summer (CANMET, 2004: 17). Increased efficacy reduces the lighting heat addition, offsetting the energy and greenhouse gas benefits of conversion to more efficient lighting sources (assuming the electricity generation mix remains the same).

A typical CFL contains 5 mg of mercury (Energy Star, 2007; Parsons, 2006: 9), a toxic substance. Without an effective recycling program it is assumed CFLs will end up in landfills, which is a potential environmental issue if CFL use increases substantially. Conversely, electricity generation, particularly coal-fired, also releases mercury into the environment. Therefore reduced demand for coal-fired electric generation could reduce the amount of atmospheric mercury emissions. These effects represent both a cost and benefit; a fair comparison of these relative effects is required to evaluate the overall environmental effect of this regulation.

### 1.1 Residential Lighting in Canada: Background

The most comprehensive data on residential lighting are provided by the 1993 Survey of Household Energy Use (SHEU) by NRCan (1994). The Canadian Residential Energy End-use Data and Analysis Centre (CREEDAC) published several studies based on the 1993 SHEU data (Fung & Ugursal, 1995; Fung & Ugursal, 1998). These data characterize a provincial indoor and outdoor residential lighting by lamp type (incandescent, halogen and fluorescent) (Fung & Ugursal, 1998: 266) as well as providing the number of incandescent lamps per room type (Fung & Ugursal, 1995: Table 3). CREEDAC's energy analysis for lighting assumes an average wattage by lamp type as well as an average duration of use. These values are 67.1 W for 2.74 hours/day for incandescent lamps (Fung & Ugursal, 1998: 265). The most recent SHEU data are from 2003 (NRCan, 2005b). These data, in published form, vary in detail from the 1993 data. For example, the 1993 data combine compact fluorescent and fluorescent tube lamps into one category whereas the 2003 data separate the two technologies. The 1993 data are presented provincially whereas the 2003 data combine Alberta, Manitoba and Saskatchewan together under the heading "Prairie." Also, the 2003 SHEU summary details the housing stock by region, type (though it excludes high-rise apartments), vintage, setting (urban/rural), income, occupation mode and household size (number of occupants). A further refinement would be to extrapolate lighting profiles (e.g., types of lighting within a dwelling, number of lamps) from the 1993 SHEU data to an updated housing stock. Other data inputs could aid this extrapolation, particularly the detailed lighting profiles found in (Jennings et al., 1996; Vorsatz et al., 1997).

Further analysis might also recognize the limitations of current CFL technology by realizing an ongoing need for incandescent lamps in certain luminaire types. For example, CFLs are ill-suited for recessed and enclosed fixtures (LRC, 2003). No literature has been found detailing the percentage of these fixture types in the Canadian residential lighting sector. The most detailed data to date on this topic is for the State of California (CEC, 1999) and the US Pacific Northwest (Jennings et al., 1996). The applicability of the US data to the Canadian market is uncertain.

Consumer surveys and laboratory tests report premature CFL failure due to rapid-cycling and other operating conditions such as elevated temperatures (LRC, 2003: 2-5; O'Rourke & Figueiro, 2001: 30). It is likely that increased use may result in a higher number of CFLs operating under less than ideal conditions. Such use could lead to a higher number of CFL failures, thereby increasing costs to the consumer due to premature replacement as well as increasing the amount of mercury disposal relative to ideal lamp operation.

## 2. METHODS

### 2.1 Overview

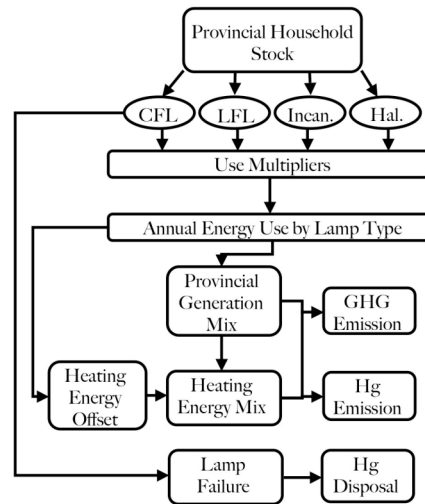
Figure 1 provides a schematic of the model<sup>2</sup> used in this analysis, of which the provincial housing stock was the foundation. This is a linear model intended for use as a feasibility tool in exploring the energy and environmental impacts of the proposed regulation. As such, it lacks non-linear feedback loops and effects that would likely exist in practice (e.g., the changing carbon intensity of the grid electricity mix as demand fluctuates).

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<sup>2</sup> The model is referred to as the "EES Model" throughout this report. This study was conducted for the Energy and Environmental Systems (EES) specialization at the University of Calgary.

The base parameter of the model is the number of Canadian residential dwelling units. This was combined with an average distribution of CFLs, linear fluorescent lamps (LFLs), basic incandescent lamps and halogen incandescent lamps (hereafter referred to as “incandescents” and “halogens,” respectively). Average wattage and daily hours of use per lamp type were used to calculate the annual energy end use by lamp type. The energy values were assessed provincially, allowing for a detailed GHG and mercury (Hg) analysis specific to provincial power generation mix. This analysis supplied the baseline for comparison when modeling the effects of the proposed NRCan regulation.

When considering the effects of the proposed regulation, the residential lighting profile was adjusted such that the decrease in incandescents was offset by an equal increase in CFLs. This affected the annual energy end use, which, in turn, affected GHG and mercury emissions. The change in emissions from a decrease in lighting energy was reduced when the heating energy offset was considered. Similarly, a reduction in cooling load during the cooling season had an effect on the energy use of conditioning systems. The heating and cooling energy offset accounted for residential heating systems and their respective fuel types on a provincial basis. Mercury emission reduction resulting from decreased electricity demand was compared with the expected increase in mercury waste resulting from lamp disposal. Emission and disposal factors were compared in terms of toxicity equivalent.



**Figure 1 - EES Model**

## 2.2 Housing Stock

In the 2003 SHEU final report, housing data for the Prairie provinces were aggregated (i.e., Alberta, Saskatchewan and Manitoba are combined under the title “Prairies”). However, NRCan provided the provincial housing data for use in this investigation (NRCan, 2008). These data are shown below in Table 1. The SHEU considered only provincial lands, meaning the northern territories were not included. It is important to note that the 2003 SHEU excluded high-rise apartments, unlike previous surveys (e.g., NRCan, 1994). High-rise apartments were therefore excluded from this analysis.

**Table 1: Number of households by Province (NRCan, 2008)**

NFLD	PEI	NS	NB	QUE	ONT	MAN	SAS	AB	BC
186,059	51,457	367,991	294,833	2,984,243	3,882,030	402,118	375,751	1,124,892	1,500,009

## 2.3 Energy Use

The electricity used for residential lighting was calculated using a method proposed by Fung and Ugursal (1995). In this method average wattages and average usage hours are stipulated for a particular lighting technology. The annual energy used per household by lighting type is calculated as follows:

$$\text{Energy used} = \text{Average wattage} \times \text{Average hours} \times \text{Number of lamps} \times 365$$

Using the above formula the aggregate energy used by all lighting types was determined, resulting in the total annual lighting energy used per household. The average wattages by lamp type and the average usage hours published by (Fung & Ugursal, 1995) were used for this analysis. However, CFLs were excluded from the aforementioned study. Generally, CFLs are used as replacements for incandescent lamps. Therefore the average usage hours of a CFL were assumed to be the same as for an incandescent lamp. The average wattage of an incandescent lamp in a Canadian house, as reported by Fung and Ugursal (1995) is 67.1 W. This analysis used the wattage of a CFL with lumen output equal to a 67.1 W incandescent lamp (i.e., 15 W using the efficacy values given by (Waide & Tanishima, 2006: 111-123) as the average wattage of a CFL used in a Canadian household (see Table 2).

**Table 2: Average Wattages and Usage Hours of Different Lighting Types**

Lighting Type	Average Wattage (W)	Average usage hours
Incandescent lamps	67.1	2.74
Halogen discharge lamps	41.1	3.80
Linear fluorescent lamps	41.1	3.80
CFLs	15.0	2.74

The lighting profile of a household was estimated using data from the 1993 and 2003 SHEU reports (NRCan, 1995, 2005b). The total number of lamps per household is reported in the 1993 SHEU on a provincial basis though similar data is not readily available for 2003. According to the 1993 SHEU, the national average of lamps per household is 27.2; for the 2003 SHEU the value is 26.4. This represents a 3% difference. Given the small difference in national lamp totals between the two surveys and the geographic detail available in 1993, the 1993 SHEU values for average number of lamps per household by province were used. The 2003 SHEU Summary reports the average distribution of lamps by lamp type as follows: 77% incandescent, 9% halogen, 5% CFL and 9% LFL (NRCan, 2005b)(NRCan, 2005b). This distribution was applied to the 1993 data on lamps per household by province to estimate the number of different types of lamps by household. The residential lighting profile used for this analysis is listed in Table 3.

**Table 3: Residential Lighting Profile**

	NFLD	PEI	NS	NB	QUE	ONT	MAN	SAS	AB	BC	Canada
Number of lamps per home	23.4	24.4	25.7	26.3	23.0	30.5	26.3	29.3	29.8	31.2	27.2
Incandescent	18.0	18.8	19.8	20.3	17.7	23.5	20.3	22.5	22.9	24.0	20.9
Halogen discharge	2.1	2.2	2.3	2.4	2.1	2.7	2.4	2.6	2.7	2.8	2.4
Linear fluorescent	2.1	2.2	2.3	2.4	2.1	2.7	2.4	2.6	2.7	2.8	2.4
CFL	1.2	1.2	1.3	1.3	1.2	1.5	1.3	1.5	1.5	1.6	1.4

The calculated electricity end use for lighting was then multiplied by the electricity transmission and distribution losses to obtain the actual electricity generation. Transmission and distribution losses depend on the transmission infrastructure and level of line loading. Values in the range of 7-12% are available in published sources (EIA, 1996; Environment Canada, 2005). In this analysis a flat value of 10% transmission and distribution losses was assumed for all the provinces (Miller & Choresl, 2004). This value was selected by considering the published average values and line loading at times of peak demand.

Lighting equivalents (NRCan, 2007a) were used to calculate the heat addition for incandescent and fluorescent lamps for both the baseline and the regulation scenarios. The corresponding effect on space conditioning systems, or energy offset, was considered. According to the results of a study conducted by the Canadian Center for Housing Technology (CANMET, 2004), 83% to 100% of lighting energy contributes to an equal heating demand reduction during the heating season. About 80% of the lighting energy internal gains contribute to cooling demand increases during the cooling season. The authors' model considered this range of heating and cooling offsets. Provincial data were used to establish the heating and cooling seasons

and the annual impact of each offset (NRCan, 2007a). To account for variations in efficiencies and GHG emissions of residential heating systems, the model included data on the provincial heating systems stock in terms of equipment efficiencies and carbon intensity of energy sources (Environment Canada, 2006; NRCan, 2003, 2006a; Statistics Canada, 2005). Provincial values of electricity carbon intensity were used for electric heating systems. To calculate cooling energy use and impacts, provincial data on number and types of air conditioning systems were combined with system energy efficiency and electricity carbon intensity by province (Environment Canada, 2006; NRCan, 2004, 2005a). A future refinement to this approach would be to adjust lighting hours of operation seasonally. As there is less daylight during the heating season it is assumed the hours of lighting used per day would be higher than in summer. This would increase the heating offset and reduce the cooling offset.

#### 2.4 Provincial Grid Related GHG Emissions

Total GHG emissions were calculated in CO<sub>2</sub> equivalent emissions using a method given by Aubé (2001). In this method the total electricity consumed is multiplied by the GHG emission intensity of the provincial electricity sector. The intensities were obtained from the Canadian National Greenhouse Gas Inventory Report (Environment Canada, 2006). The GHG emission intensities for 2003 were used to calculate the emissions for the baseline and other scenarios. The GHG emission intensity was multiplied by total energy use to obtain GHG emissions. However, for six provinces, the electricity sector GHG emission intensities were missing from 2001 onwards. Therefore, the year 2000 values were used for those six provinces. The GHG emission intensities used for the calculations are given in Table 4.

**Table 4: Provincial GHG Emission Intensities**

Province	GHG emission intensity (ktCO <sub>2</sub> eq/GWh)
Newfoundland & Labrador <sup>1</sup>	0.021
Prince Edward Island <sup>1</sup>	1.120
Nova Scotia <sup>1</sup>	0.759
New Brunswick <sup>1</sup>	0.433
Quebec <sup>2</sup>	0.002
Ontario <sup>2</sup>	0.263
Manitoba <sup>1</sup>	0.031
Saskatchewan <sup>1</sup>	0.840
Alberta <sup>2</sup>	0.848
<sup>1</sup> 2000 intensity values	
<sup>2</sup> 2003 intensity values	

#### 2.5 Mercury

The mercury analysis followed the Life Cycle Assessment (LCA) method described by Baumann and Tillman (2004) to account for increased CFL use. In this technique, an inventory analysis generates the emissions produced and the resources used during the product life cycle. In the impact assessment, the emissions and resources are related to various environmental contexts (impact categories). The impact assessment supported the overall assessment of the regulatory impact. This section describes the method used for the inventory and impact assessment of mercury emissions as part of a wider LCA.

Mercury flows to the environment due to CFL use are: deposits in landfills (due to lamp disposal) and air emissions from power plants (due to combustion). CFLs potentially increase human exposure to mercury from lamp breakage in the home. However, hazards of incidental breakage in the home were excluded from this analysis because of an inability to reasonably quantify the frequency of in-home breakage. For mercury deposition in landfills a quantity of 5 mg of mercury per bulb was used (Energy Star, 2007; Parsons, 2006: 9). The current recycling rate of 2% was used (ALMR, 2004). US EPA data show negligible mercury intensity from natural gas and oil fired power plants (Keating, 1997). Therefore, the inventory focused on coal-fired power plant emissions. To quantify the emissions from coal-fired power plants, data on mercury emission intensity (kg/TWh) of Canadian coal-fired power plants were used (CCME, 2006a: 4, 2006b: 1; CEA, 2007; Commission for Environmental Cooperation, 2007; P. J. Miller & Van Atten, 2004: 62). Mercury intensity varies significantly depending on the type of coal and power plant technology. The weighted average (by electricity generation mix) of current mercury intensity from Canadian power plants was used in calculations.

Mercury impacts are largely site dependent. Fate and transport models have been developed for local purposes (site specific), but lack applicability to assessments of large territories. The authors used the “Global Multi-media Fate, Exposure and Effects Model Uniform System for the Evaluation of Substances LCA” (USES-LCA) because it has been adapted to meet LCA-specific demands (Huijbregts et al., 2000). The main advantages of this model for application to LCA are:

- It was developed for global risk assessment purposes (previous models were originally developed for local and regional assessment)
- All relevant compartments (media of initial emission) are included in the model
- It uses revised inputs in the computation of toxicity potentials
- Worst-case values are replaced by realistic estimates

According to Toose and Mackay (2004: 4625), "a chemical (mercury), regardless of its characteristic speciation during emission into the environment, will approach the equilibrium defined by the concentration ratios observed in each medium once received into the environment". Therefore, although USES-LCA model lacks mercury speciation (Huijbregts, 2007), use for the present project should be appropriate (Mackay, 2007). The USES-LCA model provides toxicity potentials for six impact categories after initial emission to five compartments:  $TPI_{i,x,e}$  is the toxicity potential of impact category  $i$  for substance  $x$  after emission to compartment  $e$  in 1,4-dichlorobenzene (1,4-DCB) equivalents. The impact categories are:

- FAETP: freshwater aquatic ecotoxicity
- MAETP: marine aquatic ecotoxicity
- FSETP: freshwater sediment ecotoxicity,
- MSETP: marine sediment ecotoxicity,
- TETP: terrestrial ecotoxicity
- HTP: human toxicity

and the compartments are:

- air
- freshwater
- seawater
- agricultural soil
- industrial soil

The analysis considered power plant mercury emissions to be the “air” compartment, while mercury deposition in landfills were considered disposal to “industrial soil” (Baumann & Tillman, 2004; Huijbregts, 2007; Huijbregts et al., 2000).

Risk Characterization Ratios (RCRs) are the basis for calculation of toxicity potentials.  $RCR_{human,x,s,e}$  is the human risk characterization ratio of substance  $x$  at geographical scale  $s$  due to an emission to compartment  $e$  (dimensionless). Huijbregts et al. (2000: 545) explain how RCRs are calculated:

$$RCR_{human,x,s,e} = \sum_{r=1}^{r=n} \frac{PDI_{r,x,s,e}}{HLV_{r,x}}$$

$PDI_{r,x,s,e}$  is the Predicted Daily Intake via exposure route  $r$  (oral and inhalatory) of substance  $x$  for humans at geographical scale  $s$  after emission to compartment  $e$ ,  $HLV_{r,s}$  is the human limit value for exposure route  $r$  of substance  $x$ .

According to Huijbregts et al. (2000: 544), USES-LCA uses human limit values (HLVs) for all substances in order to obtain a fair comparison between substances. The above RCRs (one for each geographical scale) were aggregated using weighting factors in order to obtain a single weighted RCR for each toxicity category. For human toxicity potential, the weighting factor is the human population present at a given scale.

Huijbregts et al. (2000: 545) state, “a reference substance was used in the calculation of toxicity potentials, which follows the established use of carbon dioxide, ethylene, and chlorofluorocarbon for evaluating global warming, photochemical ozone formation, and stratospheric ozone depletion respectively. For the calculation of toxicity potentials, 1,4-DCB was taken as a reference substance”. Therefore, the calculation of the toxicity potentials was based on:

$$TP_{i,x,e} = \frac{\text{Weighted\_RCR}_{i,x,e}}{\text{Weighted\_RCR}_{ref}}$$

Human exposure and effects depend on human characteristics such as daily fish intake. Ideally, these characteristics should be implemented separately for different geographic scales (Huijbregts et al., 2000: 544). However, for practical reasons, USES-LCA uses the continental human characteristics for the global scale. Human characteristics include daily intake of drinking water, fish (assumed 90% salt-water species and 10% fresh water species), leaf crops, root crops, meat, dairy products, daily inhalation rate, body weight and daily soil particle ingestion.

## 2.6 Disposal

The method for lamp replacement assumed uniform lamp age. This was used to calculate the national lamp replacement rate by lamp type (number of replaced lamps per year in Canada by lamp type). It is important to note this basic analysis assumed a steady state condition. This implies a future circumstance after the transient period of implementation of regulatory measures. This method was used to calculate lamp replacement rates for both the baseline and regulated scenarios, limited to incandescent and compact fluorescent lamps. For each lighting technology, the following were known:

- average hours of use per day (Fung & Ugursal, 1998: 265),
- household lamp stock (see Table 3) and
- expected lamp life in hours (Waide & Tanishima, 2006: 111-123).

For each lighting technology, the number of lamps per household was multiplied by total number of households and operation hours per day for each lamp type. This value was annualized by multiplying it by 365 days per year. The annualized value was then divided by the expected lamp life, producing the national annual number of lamp failures. The lamp replacement rate was assumed to be 1:1 with lamp failure. The disposal value was equal to lamp failure.

To summarize:

$$\text{Uniform Average National Lamp Replacement} = \frac{\# \text{ households} \times \# \text{ lamps by type / household} \times \# \text{ hours of use by lamp type / day} \times 365 \text{ days / year}}{\text{expected lifetime by lamp type}}$$

Alternate disposal scenarios were considered that accounted for lamp cycling. Test data on expected lifetimes of CFLs operating under different temperature, use cycles and with different luminaires have been published by the Lighting Research Center (LRC, 2003) and Ji et al. (1999). A primary factor in premature lamp failures is frequent switching (O'Rourke & Figueiro, 2001: 30). The lifetimes of both incandescents and CFLs are affected by operating cycles, however, this study focuses on CFLs and the associated mercury toxicity impacts. CFL lifetime varies depending on CFL technology and operating cycle characteristics (O'Rourke & Figueiro, 2001: 35). Lamps are tested using a "3 hour on – 20 minute off" cycle to determine rated life (O'Rourke & Figueiro, 2001: 30). This study used a 6,000 hour lifetime for infrequently cycled lamps and a 2,000 hour for frequently cycled lamps (the latter corresponds to the "5 minutes on – 5 minutes off" test performed by O'Rourke and Figueiro (2001)). Incandescent lamps were assigned a 1,000 hour lifetime (Waide & Tanishima, 2006: 111).

## 2.7 Cost

A national average of annual lighting cost per household was calculated in the model. Any costs associated with a future regulation are projections, this analysis did not adjust for inflation or the time value of money, nor did it attempt any price forecasting. As such, the results should be considered in relative rather than absolute terms. The analysis could be improved by incorporating a more rigorous economic analysis and addressing future energy and lamp costs in greater detail.

The following outlines the household costs of the proposed NRCan regulation:

- capital cost (purchase cost of new lamps for the consumer),

- recycling cost (taken to be inclusive of lamp cost at point of purchase),
- electricity cost (associated with a switch to higher efficacy lighting) and
- space conditioning costs (associated with the space conditioning energy offset as a result of reduced heat gain from higher efficacy lighting).

Capital cost of each scenario was calculated by multiplying the annual replacement rate and the lamp cost per lamp type included in each scenario. The total capital cost is simply the summation of purchase price for each lamp type. For this calculation two inputs were used in the model: the annual replacement rate and the lamp cost (both by lamp type). Methods for calculating replacement rates are described in "Disposal" above. The cost values used for lamp types were: CDN\$3.20/lamp for CFL and CDN\$0.85/lamp for incandescent (ICF Consulting, 2008).

An effective recycling program for CFLs is desired to avoid mercury contamination in the municipal waste stream. Such a program should consist of an effective collection mechanism, CFL recycling plant, program administration body, and, at least in the initial stage, consumer awareness and education programs (Marquardt, 2002). The sum of costs of each part of the program will make the recycling cost of a lamp. The program implementation dictates who bears the cost of recycling, the consumer or the municipal district. This analysis assumes the end user pays for the recycling of the CFL at the point of purchase. CFL recycling costs are still uncertain due to the lack of large scale program experiences. Marquardt (2002) has suggested a CFL recycling cost range of CDN\$0.40-CDN\$0.80 per lamp, which is the range used in this study to estimate the household lighting cost. It is added to the CFL replacement cost.

The values for annual lighting energy use were multiplied by the national average cost of electricity (NRCan, 2006a: 43) resulting in the national consumer expenditure related to electricity use for residential lighting. Heating and cooling energy offsets are detailed above. The energy inputs associated with space conditioning offsets, including electricity, natural gas, heating oil, propane and wood, were multiplied by the respective average national costs and summed to obtain a total national cost associated with this variable. The cost values for the energy inputs were: CDN\$0.013/MJ natural gas, CDN\$0.015/MJ heating oil, CDN\$0.025/MJ propane, CDN\$0.024/MJ electricity and CDN\$0.008/MJ wood (NRCan, 2002: 14-15, 2006a). The cost of heating and cooling offsets is strictly the result of changes to household lighting technology profile and is in no way representative of the total annual household heating and cooling costs.

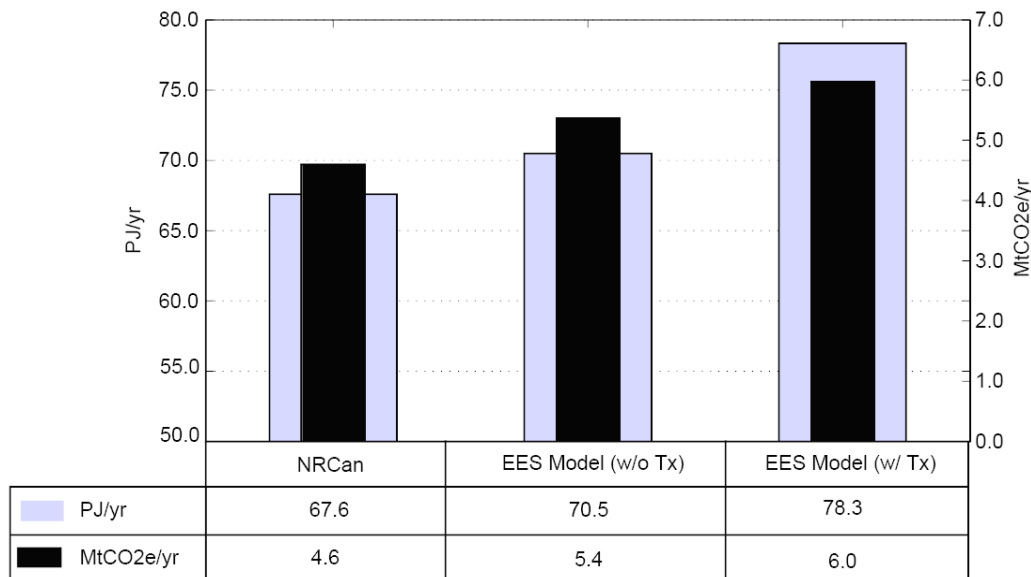
The above national costs for lighting electricity and energy offsets were summed to obtain the national total associated with the proposed NRCan regulation. This value was divided by the number of households to obtain an annual cost per household.

Cost effectiveness could be more thoroughly evaluated using annual life cycle costing (ALCC) techniques (Eley Associates, 2002; Mahlia et al., 2005). In LCC the change in life-cycle cost from the baseline to the CFL model would be calculated. The baseline could be set using available Canadian housing and lighting application data. ALCC methods published by Eley Associates (2002) would then be adopted for this calculation. The discounting factors and the time horizon for the ALCC would require further research.

### **3. RESULTS**

The model baseline used the 2003 SHEU housing stock based on a generalized lighting profile and use scenario. The results were compared with NRCan estimates in terms of energy use and GHG emissions (Figure 2). The baseline scenarios are described in Table 5.





**Figure 2: Comparison of EES Model Estimates and NRCan Data**

**Table 5: Source Descriptions for Baseline Comparison**

Source	Description/Comments
NRCan (NRCan, 2006b)	This represents the “total” energy and GHG values given online in the Energy Use Data Handbook Tables 5 and 6, respectively.
EES Model (w/o Tx)	EES results without the 10% transmission loss. This can be considered an energy end-use value.
EES Model (w/ Tx)	EES results including a 10% transmission loss. This results in larger values for energy use and GHG emissions though energy end-use values remain the same.

In comparing the EES model results with the NRCan data above, the baseline analysis revealed that:

- When transmission losses are ignored, the EES estimate is 4% higher for energy and 17% higher for GHGs than NRCan.
- When transmission losses are included in the EES model, EES is 16% higher for energy and 30% higher for GHGs than NRCan.

NRCan estimates are referred to as “end-use” values so it is assumed that transmission effects are excluded. Baseline mercury emissions were unavailable, analyses by NRCan and others excluded this. Detailed results for mercury are presented below.

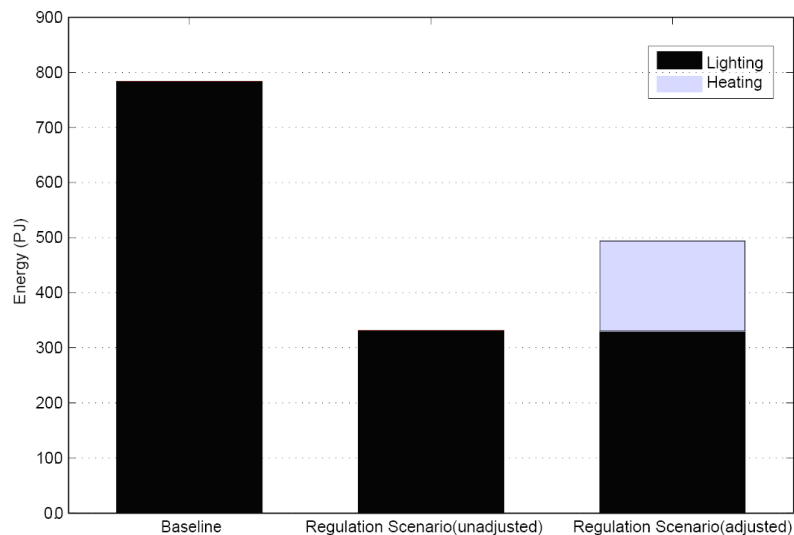
### 3.1 Regulation Scenario: Energy and Greenhouse Gas Emissions

The purpose of the analysis was to test changes in the residential lighting profile, particularly a switch from incandescent lamps to CFLs. While it would be beneficial to have the model detailed enough to allow the exclusion of certain lighting types incompatible with CFLs (i.e., recessed or enclosed luminaires, dimming or occupant sensor circuits), there are insufficient data to support this. For the primary analysis it was assumed 90% of the existing general service incandescent stock was compatible and switched to CFL though other scenarios were also considered. The 10% exclusion included those lighting types mentioned above. Nationally, as a result of this switch, lighting electricity use and greenhouse gas emissions were both reduced by 58% (reductions of 12,500 GWhr/yr and 3.5 MtCO2e/yr, respectively). This represented a 0.4% decrease in total Canadian annual GHG emissions (754 MtCO2e in 2003) (Environment Canada, 2006). It should be noted that the similarity in percent reduction for energy and GHGs was coincidental. National average results are the sum of individual provincial analyses, each having unique lighting stocks and GHG intensity factors. It should also be noted that the results presented above accounted for only the primary effects of lamp changes. When indirect space conditioning effects were considered, notably the shift of heating load from lighting to the main heating system, the figures for national energy use and GHG emission

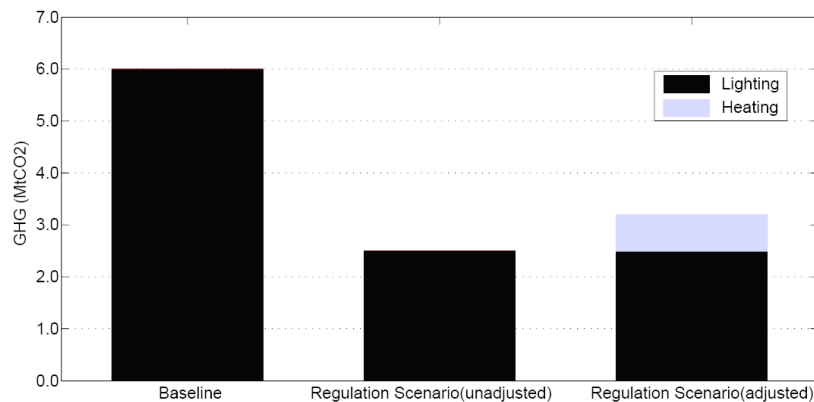
reductions changed to 37% and 46% (reductions of 8,000 GWhr/yr and 2.7 MtCO<sub>2</sub>e/yr), respectively. The absolute values for the baseline and reduction scenarios are shown in Figures 3, 4 and 5 below. Figures 3 and 4 show the scenarios including transmission losses. “Unadjusted” scenarios exclude the effects of lighting on space conditioning. “Adjusted” values include these effects. Since cooling energy savings are negligible compared to heating energy additions (less than 1% of total energy offset), the energy offset is shown as “Heating.”

Table 6 shows the annual percent change in GHG emissions and energy use on a provincial basis as a result of the regulation scenario, Figure 5 highlights provincial GHG emission values in absolute terms. While there would be a significant reduction in GHG emissions nationally, certain provinces might see an increase in GHG emissions. This is a result of the energy sources that would be used to make up a loss in heat provided by incandescent lamps, particularly in those provinces where hydroelectricity is dominant and a fraction of heating would be provided by fossil sources. However, this analysis was limited to the residential sector and it is possible that the “freed” electricity may be used to reduce greenhouse gas emissions in other sectors. All provinces experience an annual reduction in energy use.

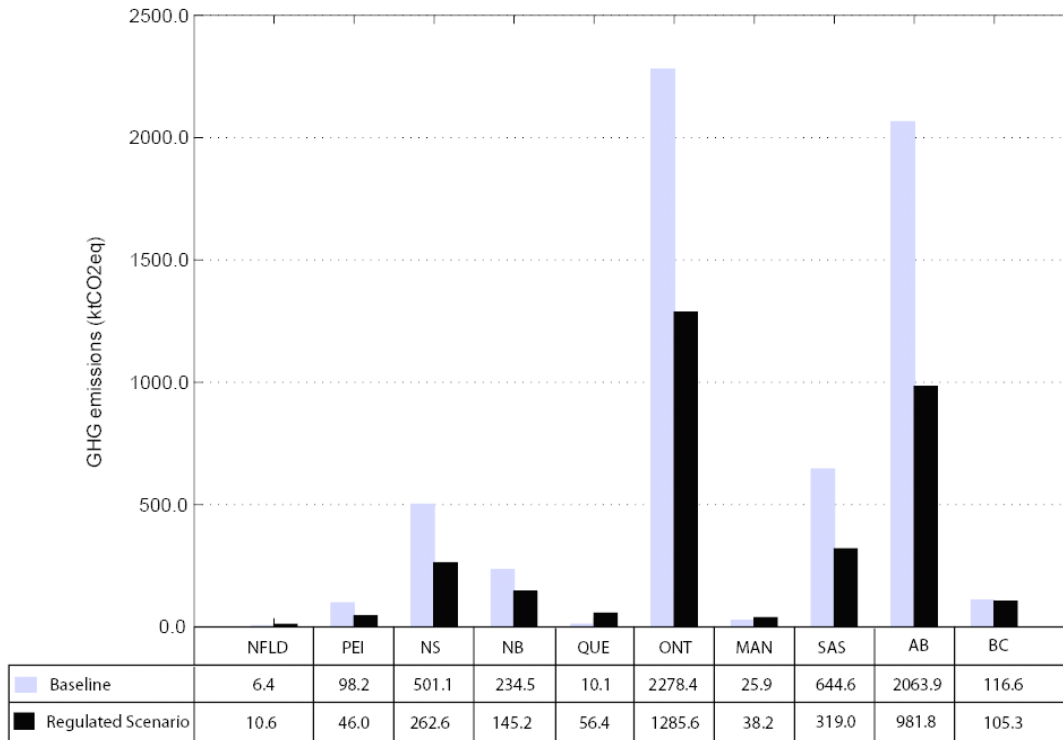
As stated above, the EES model predicted an adjusted GHG emission reduction of 2.7 MtCO<sub>2</sub>e/yr. In 2004 Canada reported emissions of 758 MtCO<sub>2</sub>e, 195 MtCO<sub>2</sub>e above its GHG emission target under the Kyoto Protocol (Environment Canada, 2006). Our analysis shows the proposed regulation could account for 1% of the emissions reduction required if Canada is to meet its Kyoto target.



**Figure 3: Energy use estimates for baseline residential lighting electricity use and conversion to CFLs**



**Figure 4: GHG emission estimates for current residential lighting energy use and conversion to CFLs**



**Figure 5: Provincial residential lighting GHG emissions (ktCO<sub>2</sub>e)**

**Table 6: Annual Percent Change in Provincial GHG and Energy Use**

	% Change in GHG	% Change in Energy Use
NFLD	65%	-33%
PEI	-53%	-34%
NS	-48%	-37%
NB	-38%	-33%
QUE	460%	-35%
ONT	-44%	-38%
MAN	48%	-29%
SAS	-51%	-29%
AB	-52%	-33%
BC	-10%	-45%

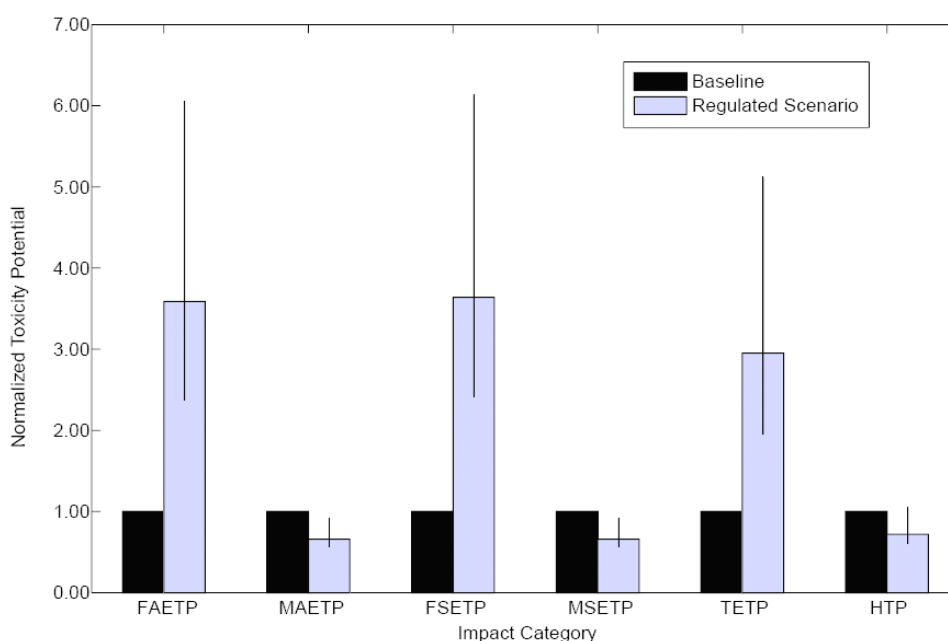
### 3.2 Regulation Scenario: Mercury

Table 7 shows annual mercury emissions and disposal values in kilograms attributable to residential lighting for both the baseline and regulated scenario. These results exclude the effects of lamp cycling and the current recycling rate (2%) is used (ALMR, 2004). National mercury emission values from coal-fired electricity generation are in the range of 2,000 kg/yr (Commission for Environmental Cooperation, 2007; P. J. Miller & Van Atten, 2004). Total mercury release for Canada is 12 T/yr (CCME, 2000).

**Table 7: Annual Mercury Emission and Disposal (kg)**

	Hg Emissions (kg)	Hg Disposal (kg)
Baseline	121	13
Regulated Scenario	54	188
Change	- 67	+ 176

Emission and disposal values should not be compared directly to judge relative benefit or harm. Mercury toxicity depends on the “compartment” in which mercury is emitted, meaning the impact of mercury emissions to air differ from those of mercury disposal. For example, according to the USES-LCA model, human toxicity potential from 1 kg of mercury emitted to the air is roughly six times more severe than the toxicity potential of the same amount of mercury deposited in a landfill. This model allows for a fair comparison of toxicity impacts between emission and disposal. Figure 6 shows the mercury toxicity potential resulting from regulation normalized against baseline values, toxicity ranges show the variability in mercury emissions from power plants<sup>3</sup>.



**Figure 6: Normalized Mercury Toxicity Potentials**

The above results show the effects by impact category of the proposed lighting regulation. The negative impacts on mercury, compared to the baseline, include a 259% increase in fresh water aquatic ecotoxicity (FAETP), a 264% increase in fresh water sediment ecotoxicity (FSETP), and a 195% increase in terrestrial ecotoxicity (TETP). On the other hand, the regulated scenario reduces human toxicity potential by 28% (HTP) and by 34% in both marine aquatic and marine sediment ecotoxicities (MAETP and MSETP, respectively). Not surprisingly, the impact categories more sensitive to combustion emissions (MAETP, MSETP and HTP) benefit from the regulatory change. Impact categories more sensitive to mercury deposition (FAETP, FSETP and TETP) are negatively affected by the same regulation.

In absolute terms, it should be noted that impacts in marine ecotoxicities are two to three orders of magnitude higher than the other impact categories. The absolute values are shown below in Table 4. The effects in marine ecotoxicities are probably because mercury circulates in the environment for long periods (Mackay, 2007), eventually accumulating in oceans. The regulation scenario would decrease the toxicity potentials in the aforementioned impact categories of the highest absolute value.

<sup>3</sup> Mercury emission intensity data are available in kgHg/GWh per power plant. However, power plant generation data in GWh/yr are not. A provincial average of mercury intensity (kgHg/GWh) was used to calculate the main results below. Error bar limits correspond to minimum and maximum mercury emission intensities

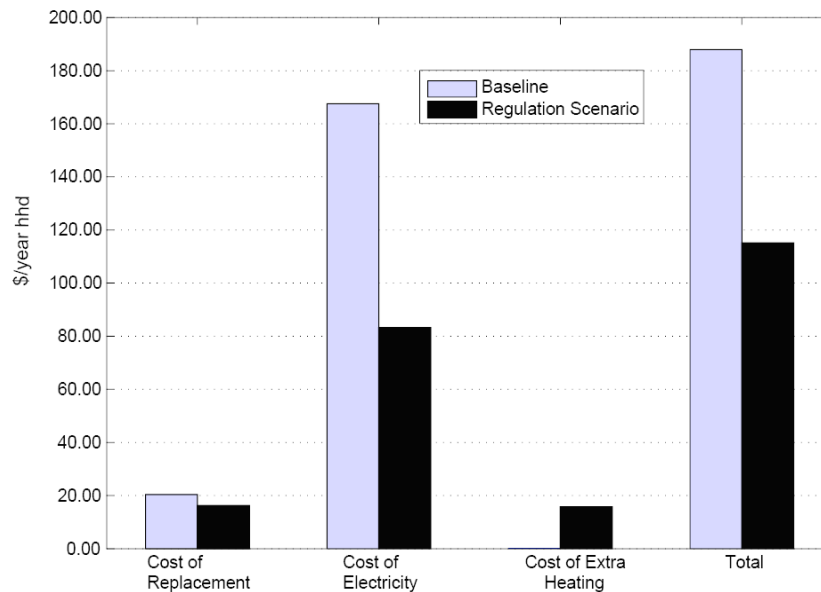
**Table 1: Toxicity by Impact Category (tonne 1,4-Dichlorobenzene eq)**

Impact Category	Baseline	Regulated scenario
FAETP	49	177
MAETP	147006	96721
FSETP	126	458
MSETP	147006	96721
TETP	4092	12058
HTP	738	531

A regulated scenario would increase the toxicity potential in three of the impact categories. However, impact categories that benefit are dominant in absolute terms. In addition, these results show that human toxicity impacts would be mitigated with wider use of CFLs. However, this analysis excludes breakage in the home, which would have much more direct effects on human health (i.e., the 28% reduction in HTP considers mercury air emissions and mercury landfilling but not lamp breakage in the home).

### 3.3 Regulation Scenario: Cost

The basic cost analysis accounts for the cost of replacement (i.e., lamp and recycling fee) and cost of electricity for both baseline and regulated scenarios. In the case of the regulated scenario, the cost of the extra heating to make up for the reduction of heat input by the lighting system was also taken into account. Figure 7 shows a rough estimate of related costs at the household level using a CDN\$0.60 point of purchase recycling fee. Annual recycling costs per household should not exceed CDN\$11.50 (adapted from (Dunmire et al., 2003)).



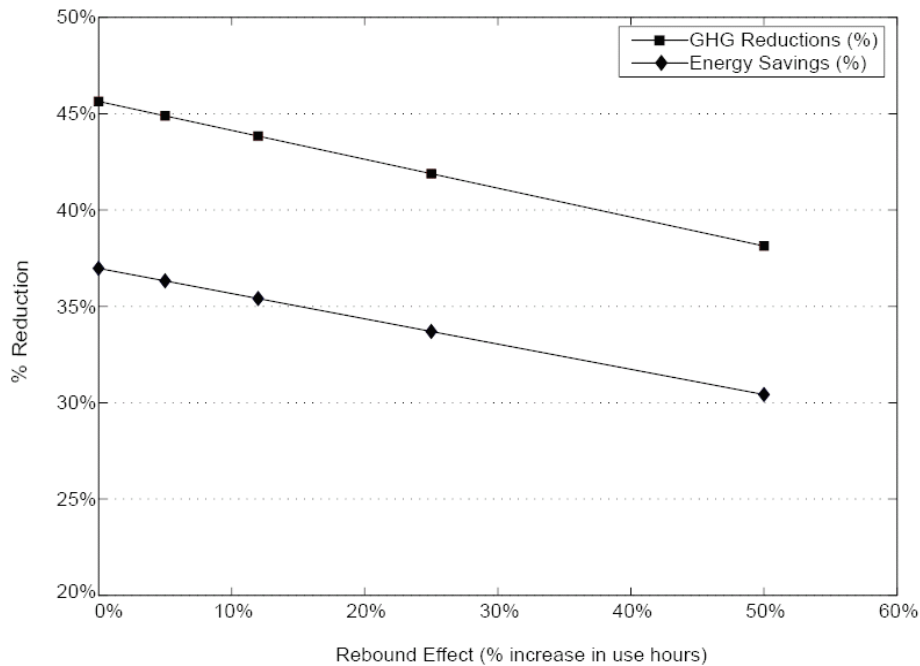
**Figure 7: Annual Cost per Household with CDN\$0.60 Recycling Fee (Average)**

Results above show the regulation would benefit the average Canadian household with an overall annual saving of about CDN\$73, which translates into CDN\$815 million when calculating the national figure. While CFLs have a higher per unit cost, their longer life results in an annualized decrease in lamp replacement costs. Households would have to bear a CDN\$16 annual increase heating cost (heating oil, natural gas, propane or wood). However, results showed the cost of electricity to be dominant in the cost analysis in both scenarios (89% in the baseline, 72% in the regulated scenario). The annual cost of electricity for lighting dropped from about CDN\$168 to CDN\$83, resulting in an CDN\$85 drop in electricity expenditures. The effects of the full range of recycling fees (i.e., CDN\$0.40 - CDN\$0.80) did not have significant economic effects at the household level due to the dominance of the cost of electricity (see Figure 11).

### 3.4 Rebound Effect

The Khazzoom-Brooks postulate states resource efficiency measures may have the effect of increasing demand for that resource (Saunders, 1992). Typical rebound effects range between 0% and 30% (Berkhout et al., 2000), although for residential lighting efficiency measures 5% to 12% can be expected (Greening et al., 2000: 394). Figure 8 below shows the rebound effect of increased lighting usage on the overall results (0% rebound corresponds to the reductions from the baseline in energy and GHG reported above).

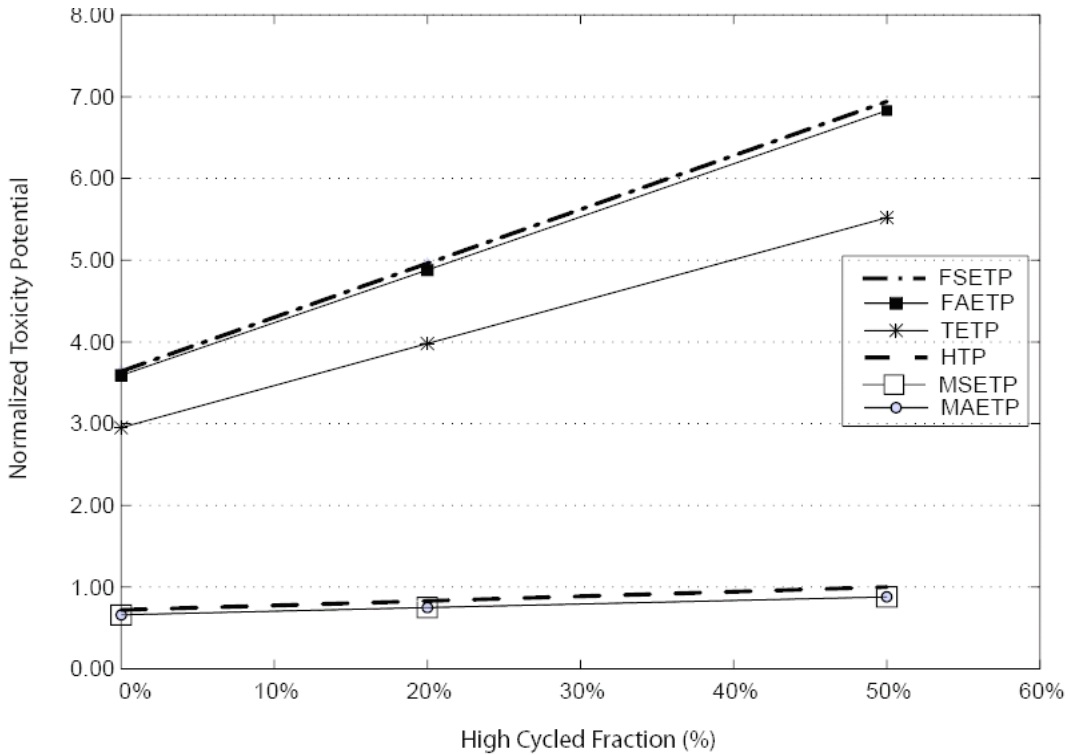
A 5% rebound effect resulted in a baseline reduction of 36% (28.5 PJ/yr) for energy use of and 45% (2.7 MtCO<sub>2</sub>e/yr) for GHG, down from 37% and 46%, respectively. Alternatively, a 12% rebound would have the effect of reducing total energy and GHG emissions by 35% (27.7 PJ/yr) and 44% (2.6 MtCO<sub>2</sub>e/yr), respectively. As seen in Figure 8, even a 50% rebound would result in significant reductions in energy use and GHG emissions. The other output variables, such as mercury toxicity potentials, unadjusted electricity cost and lamp replacement costs, increase with a rebound effect. However, the effect on the heating energy offset is slightly reduced due to the additional heat transferred from lighting.



**Figure 8: Effect of rebound effect in lighting use in terms of baseline comparison**

### 3.5 Cycling and Recycling

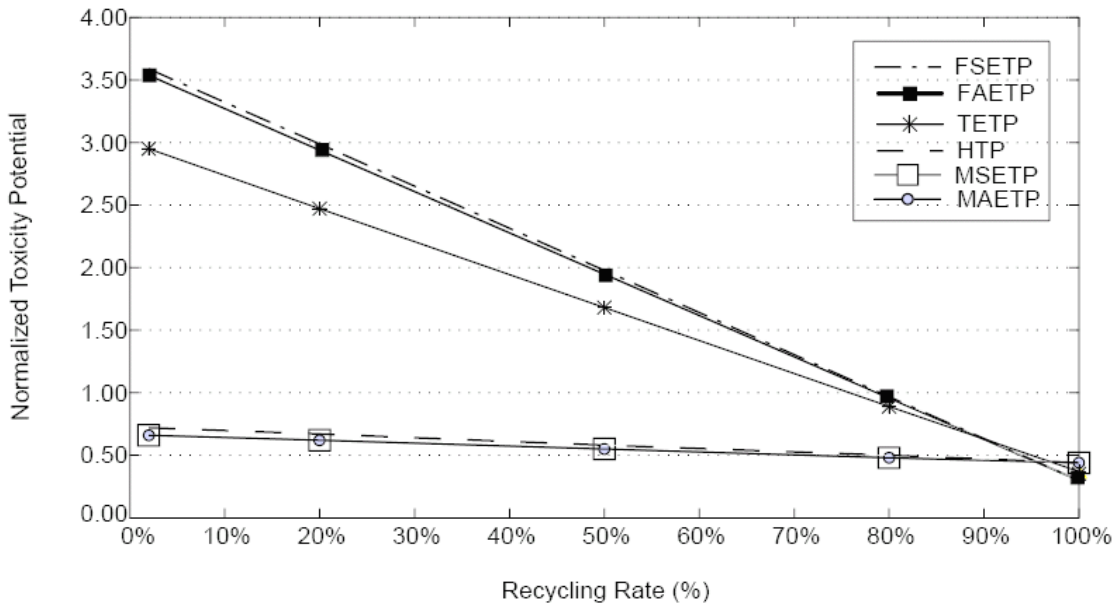
The cycling and recycling rates of CFLs affect the amount of mercury landfilled and the corresponding toxicity potentials. Figure 9 shows the change in normalized toxicity potentials as the fraction of frequently cycled CFL lamps increases. The toxicity and cost results previously reported omit the effect of frequent cycling on lamp life (this corresponds to 0% frequent cycled fraction in the figure below).



**Figure 9: Normalized mercury toxicity potentials vs. frequently cycled fraction of CFL**

Results show an increase in all toxicity potentials when the fraction of frequent cycling CFL lamps increases. The local impact categories (FSETP, FAETP, and TETP) show steep slopes, indicating these are the most sensitive to mercury cycling (due to increased lamp disposal). It should be noted that the positive effect on toxicity potentials in global impact categories resulting from the regulation (HTP, MAETP, and MSETP) become negative with a fraction of frequent cycling CFL lamps around 50%.

Similarly, Figure 10 shows the change in normalized toxicity potentials with an increase in the recycling rate. The toxicity results previously presented considered the current recycling rate (2%) (ALMR, 2004).

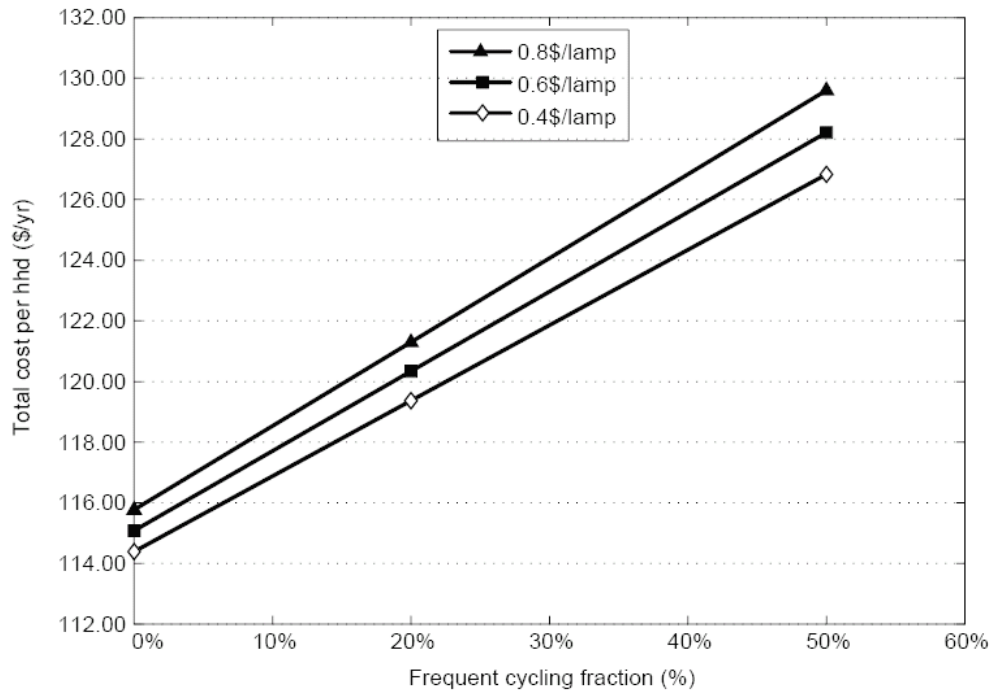


**Figure 10: Normalized mercury toxicity potentials vs. recycling rate (CFL)**

Similarly to the frequent cycling effect, results show the local impact categories (FSETP, FAETP, and HTP) are most sensitive to changes in the recycling rate. Figure 10 indicates that any recycling program would have to be at least 80% effective (meaning an 80% take back rate of lamps used) to achieve a net

benefit in toxicity potential for all impact categories. In a more general perspective, the results shown in Figure 10 could be interpreted as mercury toxicity potentials versus any reduction in mercury flow to the ground. Therefore, similar results would be achieved if the X-axis represented “assured landfill impermeability” or “mercury content reduction in CFLs (assuming constant efficacy)”.

The effect of frequent cycling on annualized cost is shown in Figure 11 and shows the variability caused by the range of recycling rates considered. A 50% frequent cycling scenario results in approximately a 10% annual cost increase due to increased lamp replacement.



**Figure 11: Effect of frequent cycling on total cost per household**

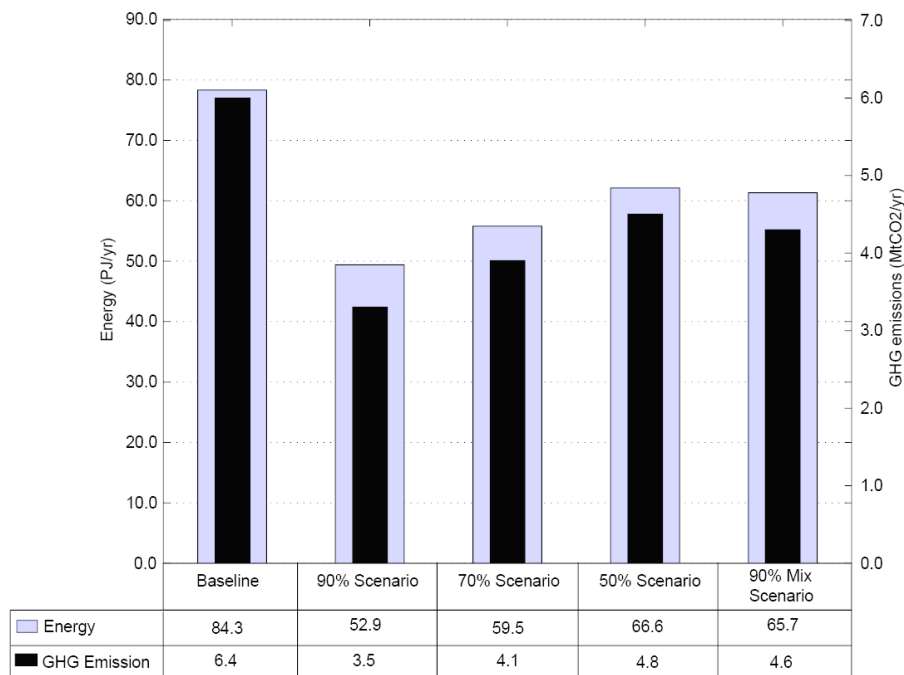
### 3.6 Alternative Use Scenarios

The results and analysis presented above assumed 90% of the existing incandescent stock were compatible and switched to CFL. However, the actual transition may be different from the assumed fraction. The actual residential lighting profile may have a high proportion of ill-suited operating conditions, such as dimmable circuits or occupancy sensors. Also, users may seek alternatives to CFLs. Halogen lamps can meet the efficacy levels set by the Tier 1 NRCan standard and can be used in applications where CFLs are ill-suited. Therefore, three other scenarios were tested for total energy use and GHG emissions. The scenarios tested are as follows:

- 70% scenario: 70% of the existing incandescent stock replaced by CFLs.
- 50% scenario: 50% of the existing incandescent stock replaced by CFLs.
- 90% mix scenario: 50% of the incandescent stock replaced with CFLs and 40% of the incandescent stock is replaced with halogen lamps (a total of 90% of the incandescent stock is replaced).

The total energy used and GHG emissions under all scenarios tested are shown in Figure 12. These results report adjusted values (inclusive of energy offset).





**Figure 12: Energy use and GHG emissions under different scenarios**

As expected, the total energy saved and the GHG emission offset decrease with the decreasing fraction of incandescent stock replaced with efficient lighting technologies. However, even a 50% transition from incandescent lamps to CFLs would reduce the total energy used by 16 PJ/year (21% reduction) and GHG emission by 1.5 MtCO<sub>2</sub>e/year (26% reduction). Furthermore, it is imperative to recognize that, although lower in efficacy than CFLs, use of other efficient lighting sources such as halogen lamps may achieve significant energy and GHG emission reductions. For example, under the 90% mix scenario described above, which considered a significant halogen lamp fraction in the residential lighting stock, the total energy use decreased by 17 PJ/year (22% reduction) and the GHG emission dropped by 1.7 MtCO<sub>2</sub>e/year (28% reduction).

## 4. CONCLUSIONS AND RECOMMENDATIONS

### 4.1 Conclusions

Implementation of the NRCan regulation with CFLs as the dominant replacement technology would generate significant reductions in total Canadian residential lighting energy demand (about 37%) and greenhouse gas emissions (about 46%). Considering only the effects of residential lighting and space conditioning, certain provinces with low carbon intensive electricity generation mix may experience an increase in residential greenhouse gas emissions (up to 460% in Quebec) as a result of offsetting the heat contribution of lighting with a more carbon-intensive heating fuel. On a broader basis, the “freed” electricity might reduce GHG emissions in other sectors within these provinces. The relative GHG increases in these provinces are notable; however, they are small in absolute terms compared with declines in other provinces. Therefore, the negative impacts for these provinces are diluted in the national average which is dominated by GHG emissions in more polluting provinces such as Ontario and Alberta.

The average household would experience an annual reduction in costs associated with lighting though some would experience a lesser increase in fuel costs for heating. Annual lamp replacement cost as compared to the baseline varies with cycling effect. With infrequent cycling there is a slight reduction in annual replacement costs; a 50% frequent-cycling effect produces a 10% annualized cost increase. However, the cost analysis is clearly dominated by the cost of electricity for lighting, which provides large reductions in the regulated scenario. Therefore, the overall costs associated with lighting (including heating offset) in the regulated scenario would be significantly lower than the current ones. The overall annual savings (calculated to be about CDN\$73 per household or CDN\$815 million nationally) show the regulation can be cost-effective at the household level.

The regulated scenario would reduce the mercury air emissions by 67 kg/yr, and increase the landfill disposal by 176 kg/yr, which represents a small fraction of the total national mercury emission of 12 T/yr

(CCME, 2000). Our mercury toxicity analysis showed mixed results, with some impact categories better and other worse. Marine aquatic and sediment ecotoxicities benefit from the regulated lighting scenario. However, more localized impact categories (terrestrial, fresh water aquatic and sediment ecotoxicities) could be negatively affected because of CFL disposal. This increase in toxicity potential is significant. A greater benefit would certainly be realized by implementing an effective recycling program for CFLs. The model output shows a 28% reduction in human toxicity potential with the regulated lighting scenario. However, this analysis excludes toxicity impacts of lamp breakage in the home. Both frequent cycling and recycling rates affect toxicity potentials, especially local impact categories. When about 50% of the lamps are frequently cycled CFLs, impact in categories previously benefitted by the regulatory change become negative. On the other hand, an 80% recycling capture rate would be required to achieve a net benefit in all impact categories.

Other scenarios with lower CFL penetration rate would result in lower, but still significant, reductions in energy use and GHG emissions. Switching 50% of currently installed incandescents to CFLs would result in 21% and 26% reductions in energy use and GHG emissions, respectively.

#### 4.2 Recommendations

The accuracy of the results presented here could be improved through the development of a more detailed residential lighting profile. Future residential energy use surveys conducted at the national level could include detailed questions to create such a profile.

If it is the intent of this regulation to reduce Canada's national GHG emissions, this analysis has found the regulation could be a cost attractive means of doing so at the consumer level. Administrative and recycling costs require additional investigation. This study found that with a high penetration (90%) of CFLs in the marketplace, the proposed regulation could achieve 1% of Canada's obligation under the Kyoto protocol. A reduction in grid power use for lighting is a further benefit.

The toxicity potentials in the local impact categories (FAETP, FSETP and TETP) increase significantly due to CFL disposal. Removing mercury from the waste stream would reduce these negative impacts. As a result, it is recommended that a widespread lamp recycling program be developed in conjunction with the proposed lighting standard. Development and operational expenditures of the recycling program might take advantage of the reduction in annual lighting electricity costs. No effort was made here to cost the positive and negative mercury toxicity impacts. Policy analysts and legislators would benefit from such work.

#### REFERENCES

- ALMR. (2004). *National Mercury-Lamp Recycling Rate and Availability of Lamp recycling Services in the U.S.* Association of Lighting and Mercury Recyclers. Retrieved December 2007, from [http://www.nema.org/lamprecycle/docs/ALMR\\_capacity\\_statement.pdf](http://www.nema.org/lamprecycle/docs/ALMR_capacity_statement.pdf).
- Aubé, F. (2001). *Guide for Computing CO<sub>2</sub> Emissions Related to Energy Use.* CANMET Energy Diversification Research Laboratory. Retrieved October 10, 2007, from <http://cetc-varenes.nrcan.gc.ca/fichier.php/codectec/En/2001-66/2001-66e.pdf>.
- Baumann, H., & Tillman, A. M. (2004). *The Hitchhiker's Guide to LCA.* Gothenburg: Environmental Systems Analysis, Chalmers University of Technology
- Berkhout, P. H. G., Muskens, J. C., & W. Velthuisen, J. (2000). Defining the rebound effect. *Energy Policy*, 28(6-7), 425-432.
- CANMET. (2004). *Benchmarking of Energy Savings Associated with Energy Efficient Lighting in Houses.* CANMET Energy Technology Centre.
- CCME. (2000). *Canada-Wide Standards for Mercury Emissions.* Retrieved November, 2007, from [www.ccme.ca/assets/pdf/mercury\\_emis\\_std\\_e1.pdf](http://www.ccme.ca/assets/pdf/mercury_emis_std_e1.pdf).
- CCME. (2006a). *Canada-Wide Standards for Mercury Emissions from Coal-Fired Electric Power Generation Plants.* Retrieved October 10, 2007, from [http://www.ccme.ca/assets/pdf/hg\\_epg\\_cws\\_w\\_annex.pdf](http://www.ccme.ca/assets/pdf/hg_epg_cws_w_annex.pdf)
- CCME. (2006b). *Data Analysis in Support of the Developmebnt of a Canada-Wide Standards for Mercury Emissions from Coal-Fired Electric Power Generation Plants.* Canadian Council of Ministers of the Environment. Retrieved October 10, 2007, from [http://www.ccme.ca/assets/pdf/hg\\_data\\_analysis\\_e.pdf](http://www.ccme.ca/assets/pdf/hg_data_analysis_e.pdf).
- CEA. (2006). *Power Generation in Canada - A Guide.* Canadian Electricity Association. Retrieved October 4, 2007, from <http://www.canelect.ca/en/Pdfs/HandBook.pdf>
- CEA. (2007). *Canadian Electricity Association Mercury Program.* Retrieved October 18, 2007, from [http://www.ceamercuryprogram.ca/EN/mercury\\_home.html](http://www.ceamercuryprogram.ca/EN/mercury_home.html)

- CEC. (1999). *Lighting Efficiency Technology Report: Volume 1: California Baseline* California Energy Commission. Retrieved October 02, 2004, from <http://www.energy.ca.gov/efficiency/lighting/VOLUME01.PDF>.
- Commission for Environmental Cooperation. (2007). Canada 2002 power plant emissions. Retrieved October 18, 2007, from [http://www.cec.org/pubs\\_docs/documents/index.cfm?id=1688](http://www.cec.org/pubs_docs/documents/index.cfm?id=1688)
- Dunmire, C., Calwell, C., Jacob, A., Ton, M., Reeder, T., & Fulbright, V. (2003). *Mercury in fluorescent lamps: environmental consequences and policy implications for NRDC*. Ecos Consulting. Retrieved November, 2007, from <http://www.bvsde.paho.org/bvsaiia/fulltext/NRDCMercury.pdf>.
- EIA. (1996). *Residential Lighting: Use and Potential Savings*. U.S. Energy Information Administration. Retrieved October 10, 2007, from <http://www.eia.doe.gov/emeu/lighting/contents.html>.
- Eley Associates. (2002). *Life Cycle Cost Methodology - 2005 California Building Energy Efficiency Standards*. California Energy Commission. Retrieved October 02, 2007, from [http://www.energy.ca.gov/title24/2005standards/archive/documents/2002-04-02\\_workshop/2002-03-20\\_LIFE\\_CYCLE.PDF](http://www.energy.ca.gov/title24/2005standards/archive/documents/2002-04-02_workshop/2002-03-20_LIFE_CYCLE.PDF).
- Energy Star. (2007). Frequently Asked Questions Information on Compact Fluorescent Light Bulbs (CFLs) and Mercury August 2007. Retrieved September 28, 2007, from [http://www.energystar.gov/ia/partners/promotions/change\\_light/downloads/Fact\\_Sheet\\_Mercury.pdf](http://www.energystar.gov/ia/partners/promotions/change_light/downloads/Fact_Sheet_Mercury.pdf)
- Environment Canada. (2005). *Report of the Development of a Canadian Electricity Sector Module for the Integrated Planning Model®*. Retrieved November, 2007, from [http://www.ec.gc.ca/cleanair-airpur/caol/canus/IPM\\_TECHNICAL/ipm\\_technical\\_report/toc\\_e.cfm](http://www.ec.gc.ca/cleanair-airpur/caol/canus/IPM_TECHNICAL/ipm_technical_report/toc_e.cfm).
- Environment Canada. (2006). *National Inventory Report, 1990-2004 - Greenhouse Gas Sources and Sinks in Canada*. Retrieved October 18, 2007, from [http://www.ec.gc.ca/pdb/ghg/inventory\\_report/2004\\_report/toc\\_e.cfm](http://www.ec.gc.ca/pdb/ghg/inventory_report/2004_report/toc_e.cfm).
- Fung, A. S., & Ugursal, V. I. (1995). *Residential Lighting Energy Use in Canada - Final Report*. Canadian Residential Energy End-use Data and Analysis Centre.
- Fung, A. S., & Ugursal, V. I. (1998). Residential Lighting Energy Use in Canada. *Energy Studies Review*, 8(3), 264-268.
- Greening, L. A., Greene, D. L., & Difiglio, C. (2000). Energy efficiency and consumption—the rebound effect—a survey. *Energy Policy*, 28(6-7), 389-401.
- Huijbregts, M. A. J. (2007). Personal Correspondence.
- Huijbregts, M. A. J., Thissen, U., Guinee, J. B., Jager, T., Kalf, D., van de Meent, D., et al. (2000). Priority Assessment of Toxic Substances in Life Cycle Assessment. Part I: Calculation of Toxicity Potentials for 181 Substances with the Nested Multi-Media Fate, Exposure and Effects Model USES-LCA. *Chemosphere*, 41(4), 541-573.
- ICF Consulting. (2008). *General Service Lighting*. ICF Consulting Canada, Inc.
- Jennings, J., Moezzi, M., Brown, R., Mills, E., Sardinsky, R., Heckendorn, B., et al. (1996). *Residential Lighting: The Data to Date*. Lawrence Berkeley National Laboratory. Retrieved October 4, 2007, from <http://gaia.lbl.gov/btech/papers/38454.pdf>.
- Ji, Y., Davis, R., & Chen, W. (1999). An Investigation of the Effect of Operating Cycles on the Life of Compact Fluorescent Lamps. *Journal of the Illuminating Engineering Society*, 28(2), 57-62.
- Keating, M. H. (1997). *Mercury study report to Congress. Volume 2. An inventory of anthropogenic mercury emissions in the United States*. Office of Air Quality Planning and Standards, U.S. Environmental Protection Agency. Retrieved October 02, 2007, from <http://www.epa.gov/ttncaaa1/t3/reports/volume2.pdf>.
- LRC. (2003). *Durability Testing for ENERGY STAR® Residential Light Fixtures*. Lighting Research Center, Rensselaer Polytechnic Institute. Retrieved September 20, 2007, from <http://www.lrc.rpi.edu/programs/lightingTransformation/pdf/durabilityTestingFinalReport.pdf>.
- Mackay, D. (2007). Personal Correspondence.
- Mahlia, T. M. I., Said, M. F. M., Masjuki, H. H., & Tamjis, M. R. (2005). Cost-benefit Analysis and Emission Reduction of Lighting Retrofits in Residential Sector. *Energy and Buildings*, 37(6), 573-578.
- Miller, D., & Chorel, M. (2004). *A Study on the Efficiency of Alberta's Electrical Supply System*. JEM Energy & Associates. Retrieved July 2008, from [http://www.casahome.org/wp-content/uploads/2006/10/EEC\\_JEM%20Final%20Report13Oct2004.pdf](http://www.casahome.org/wp-content/uploads/2006/10/EEC_JEM%20Final%20Report13Oct2004.pdf).
- Miller, P. J., & Van Atten, C. (2004). *North American Power Plant Air Emissions*. Commission for Environmental Cooperation of North America. Retrieved October 10, 2007, from [http://www.cec.org/files/PDF/POLLUTANTS/PowerPlant\\_AirEmission\\_en.pdf](http://www.cec.org/files/PDF/POLLUTANTS/PowerPlant_AirEmission_en.pdf)

- NRCan. (1994). *1993 Survey of Household Energy Use: National Results*. Natural Resources Canada's Office of Energy Efficiency. Retrieved October 10, 2007, from [http://oee.nrcan.gc.ca/infosource/pdfs/EnergyUseNational-full\\_e.pdf](http://oee.nrcan.gc.ca/infosource/pdfs/EnergyUseNational-full_e.pdf).
- NRCan. (1995). *1993 Survey of Household Energy Use: Provincial Results Retrieved*. Natural Resources Canada's Office of Energy Efficiency. Retrieved October 10, 2007, from <http://oee.nrcan.gc.ca/infosource/pdfs/EnergyUseProvincial-full.pdf>.
- NRCan. (2002). *A Guide to Residential Wood Heating*. Retrieved November 08, 2007, from [http://www.canren.gc.ca/prod\\_serv/index.asp?CaId=103&PgId=576](http://www.canren.gc.ca/prod_serv/index.asp?CaId=103&PgId=576).
- NRCan. (2003). *Heating with Electricity*. Retrieved November 10, 2007, from [http://oee.nrcan.gc.ca/publications/infosource/pub/home/heating\\_with\\_electricity\\_Content.cfm?text=N&printview=N](http://oee.nrcan.gc.ca/publications/infosource/pub/home/heating_with_electricity_Content.cfm?text=N&printview=N).
- NRCan. (2004). *Air conditioning your home*. Retrieved November 08, 2007, from [http://www.oee.nrcan.gc.ca/publications/infosource/pub/energy\\_use/air-conditioning-home2004/air-conditioning.pdf](http://www.oee.nrcan.gc.ca/publications/infosource/pub/energy_use/air-conditioning-home2004/air-conditioning.pdf).
- NRCan. (2005a). *Survey of Household Energy Use 2003*. Natural Resources Canada's Office of Energy Efficiency. Retrieved September 20, 2007, from [http://www.oee.nrcan.gc.ca/corporate/statistics/neud/dpa/data\\_e/sheu03/tables.cfm?attr=0](http://www.oee.nrcan.gc.ca/corporate/statistics/neud/dpa/data_e/sheu03/tables.cfm?attr=0).
- NRCan. (2005b). *Survey of Household Energy Use 2003 (Summary Report)*. Natural Resources Canada's Office of Energy Efficiency. Retrieved September 20, 2007, from <http://www.oee.nrcan.gc.ca/Publications/statistics/sheu-summary/index.cfm>.
- NRCan. (2006a). *Energy Use Data Handbook 1990 and 1998 to 2004*. Office of Energy Efficiency, Natural Resources Canada. Retrieved October 10, 2007, from <http://oee.nrcan.gc.ca/Publications/statistics/handbook06/index.cfm>.
- NRCan. (2006b). *Energy Use Data Handbook Tables (Canada)*. Retrieved October 18, 2007, from [http://oee.nrcan.gc.ca/corporate/statistics/neud/dpa/handbook\\_res\\_ca.cfm?attr](http://oee.nrcan.gc.ca/corporate/statistics/neud/dpa/handbook_res_ca.cfm?attr).
- NRCan (2007a). Personal Correspondence.
- NRCan. (2007b). *Lighting the Way to a Greener Future: Canada's New Government to Ban Inefficient Light Bulbs*. Retrieved September 28, 2007, from [http://www.nrcan-nrcan.gc.ca/media/newsreleases/2007/200735\\_e.htm](http://www.nrcan-nrcan.gc.ca/media/newsreleases/2007/200735_e.htm).
- NRCan (2008). Personal Correspondence.
- O'Rourke, C., & Figueiro, M. (2001). Long-Term Performance of Screwbase Compact Fluorescent Lamps. *Journal of the Illuminating Engineering Society*, 30(2), 30-39.
- Parsons, D. (2006). The Environmental Impact of Compact Fluorescent Lamps and Incandescent Lamps for Australian Conditions. *The Environmental Engineer*, 7(2), 8-14.
- Pfeifer, R. P. (1996). Comparison Between Filament Lamps and Compact Fluorescent Lamps. *International Journal of Life Cycle Assessment*, 1(1), 8-14.
- Saunders, H. D. (1992). The Khazzoom-Brookes postulate and neoclassical growth. *The Energy Journal*, 13(4), 131-148.
- Statistics Canada. (2005). *Spending Patterns in Canada*. Retrieved November 01, 2007, from <http://www.statcan.ca/bsolc/english/bsolc?catno=62-202-XIE#formatdisp>.
- Toose, L. K., & Mackay, D. (2004). Adaptation of fugacity models to treat speciating chemicals with constant species concentration ratios. *Environmental science & technology*, 38(17), 4619-4626.
- Vorsatz, D., Shown, L., Koomey, J., Moezzi, M., Denver, A., & Atkinson, B. (1997). *Lighting Market Sourcebook for the U.S.* Environmental Energy Technologies Division, Lawrence Berkeley National Laboratory. Retrieved October 10, 2007, from <http://enduse.lbl.gov/Info/LBNL-39102.pdf>.
- Waide, P., & Tanishima, S. (2006). *Light's Labour's Lost : Policies for Energy-Efficient Lighting*. Paris: OECD/IEA.